

U. S. Fish and Wildlife Service
Region 2

**CONTAMINANTS INVESTIGATION AT GRULLA NATIONAL
WILDLIFE REFUGE,
ROOSEVELT COUNTY, NEW MEXICO 2003
Project ID No. 1261-2N49**



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ABSTRACT

In August, 2003, a contaminants investigation was initiated at Grulla National Wildlife Refuge (Grulla NWR) by U.S. Fish and Wildlife Service personnel. The purpose of this investigation was to measure contaminant levels in the southwest portion of Grulla NWR to determine if contamination was migrating from the Arch Landfill into the Refuge, as well as determine the ecological significance of these contaminants. Surficial soils were collected from Grulla NWR and analyzed for metals, organochlorine pesticides, total polychlorinated biphenyls (PCBs), and total petroleum hydrocarbons (TPH). The resulting data were compared to baseline data collected during the same time frame from Muleshoe National Wildlife Refuge (Muleshoe NWR) and with available ecological screening criteria.

The results of the metals analyses indicate that metals are present in surficial soils at higher levels at Muleshoe NWR than at Grulla NWR. Certain metals, primarily the metalloid boron, exceed ecological screening criteria at Muleshoe NWR, but this may be attributed to the natural alkaline soils at the refuge. The results of the organic analyses indicate that residual organochlorine pesticides, total PCBs, and TPH are present in surficial soils at very low levels at both refuges. As with the metals, the organics detected at Grulla NWR were lower than the levels measured at Muleshoe NWR. Overall, the contaminants detected at both refuges were below concentrations where adverse affects to ecological resources would be expected to occur.

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TABLE OF CONTENTS

INTRODUCTION	1
STUDY AREA & BACKGROUND	1
MATERIALS & METHODS	3
RESULTS & DISCUSSION	7
Metals	9
Organochlorine Pesticides	25
Polychlorinated Biphenyls	30
Petroleum Hydrocarbons	31
Macroinvertebrates	33
CONCLUSIONS & RECOMMENDATIONS	33
REFERENCES	36
ANALYTICAL METHODS	Appendix A

LIST OF FIGURES

Figure 1. General Locations of Grulla and Muleshoe National Wildlife Refuges	2
Figure 2. Sampling Points at Grulla National Wildlife Refuge, 2003	5
Figure 3. Sampling Points at Muleshoe National Wildlife Refuge, 2003	6

LIST OF TABLES

Table 1. Location of sample sites at Grulla National Wildlife Refuge and Muleshoe National Wildlife Refuge, 2003	4
Table 2. Moisture, sand, silt, and clay content as percentages measured in soil/sediment samples collected from six sites at Grulla Lake National Wildlife Refuge and six sites at Muleshoe National Wildlife Refuge, 2003	8
Table 3. Results of metals analysis in mg/kg dry weight for soil samples collected from six sites at Grulla Lake National Wildlife Refuge, Roosevelt County, New Mexico, 2003	10
Table 4. Results of metals analysis in mg/kg dry weight for soil/sediment samples collected from six sites at Muleshoe Lake National Wildlife Refuge, Bailey County, Texas, 2003	11
Table 5. Results of organochlorine pesticide analyses in mg/kg dry weight for six soil samples collected from Grulla National Wildlife Refuge, Roosevelt County, New Mexico, 2003	26
Table 6. Results of organochlorine pesticide analyses in mg/kg dry weight for six soil/sediment samples collected from Muleshoe National Wildlife Refuge, Bailey County, Texas, 2003	27
Table 7. Results of total polychlorinated biphenyl and total petroleum hydrocarbon analyses in mg/kg dry weight for soil samples collected from six sites at Grulla National Wildlife Refuge and soil/sediment samples collected from six sites at Muleshoe National Wildlife Refuge, 2003	30
Table 8. Macroinvertebrate taxa and their associated trophic relations collected from Upper Paul's Lake, Muleshoe National Wildlife Refuge, 2003	34
Table 9. Number and percentage trophic groups of macroinvertebrate taxa represented in samples collected from Upper Paul's Lake, Muleshoe National Wildlife Refuge, 2003.	34

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INTRODUCTION

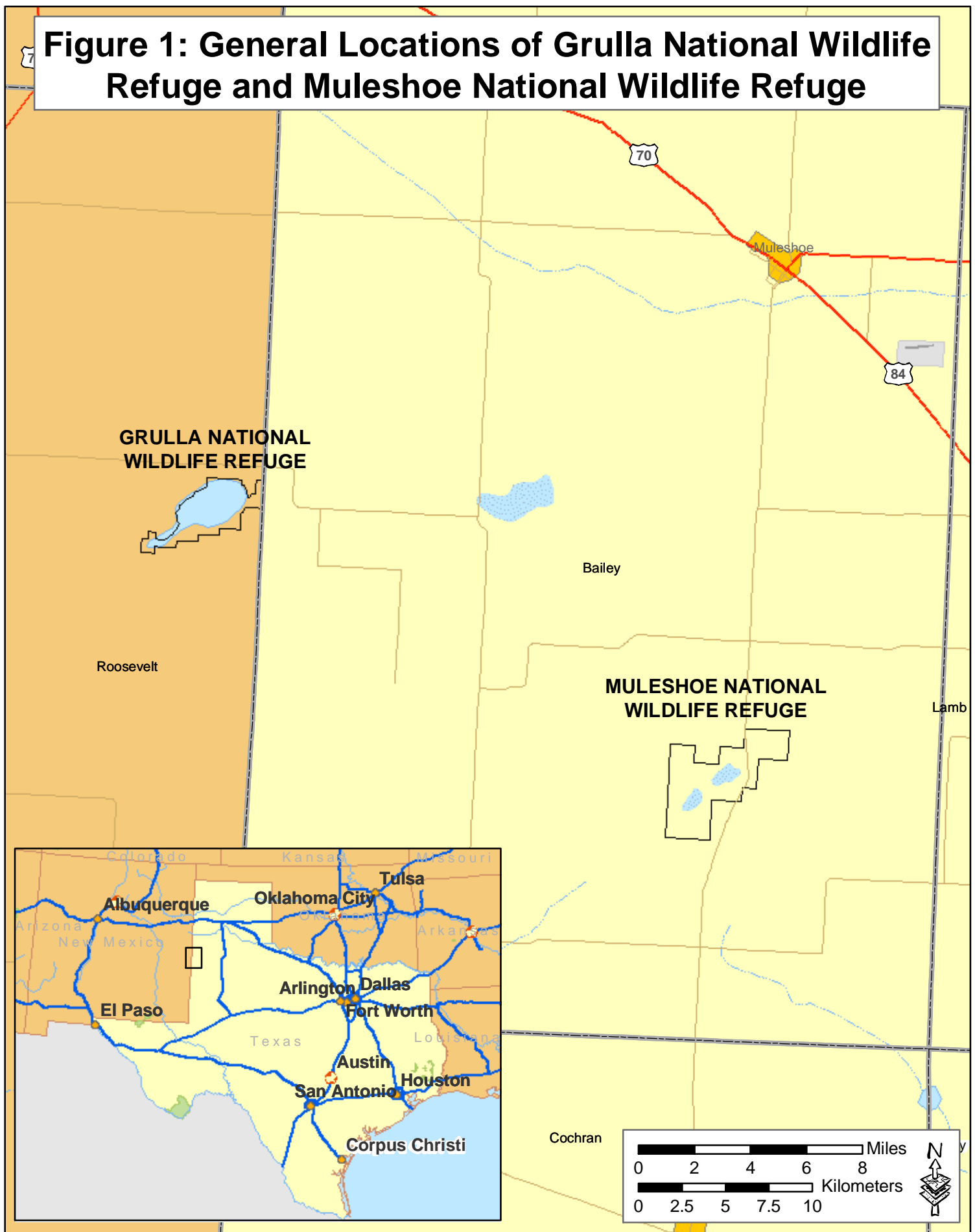
In August, 2003, a contaminants investigation was initiated at Grulla National Wildlife Refuge (Grulla NWR) by U.S. Fish and Wildlife Service (USFWS) personnel from the Arlington, Texas Ecological Services Field Office (ESFO). The purpose of this investigation was to measure contaminant levels in the southwest portion of Grulla NWR to determine if contamination was migrating from the Arch Landfill into the Refuge, and if so, determine the ecological significance of those contaminants. Surficial soils were collected from Grulla NWR and analyzed for metals, organochlorine pesticides, total polychlorinated biphenyls, and total petroleum hydrocarbons. The resulting data were compared to baseline data collected during the same time frame from Muleshoe National Wildlife Refuge (Muleshoe NWR) and from available ecological screening criteria.

STUDY AREA & BACKGROUND

Established as a refuge in 1969, Grulla NWR is located within the southern high plains in the Edwards Plateau Ecosystem near the town of Arch in far eastern Roosevelt County, New Mexico (Figure 1). The refuge encompasses 3,236 acres (1310 hectares), including a saline lake bed known as Salt Lake which composes approximately 66% [2,160 acres (874 hectares)] of the refuge's total acreage. The remaining acreage within the refuge is managed as grasslands. The soils surrounding the lake bed are within the Potter-Mansker association and are calcareous, with a high lime content (USDA, 1967). The climate of this area is considered semi-arid. Ambient air temperatures average 21.2°Fahrenheit (F) [-5.9°Celsius (C)] in the winter and 91.7°F (33.2°C) in the summer. Winds are predominantly out of the south-southwest (NWS, personal communication, 1998). Average annual rainfall is approximately 16 inches (41 centimeters) (NMWQCC, 1994).

Located immediately up gradient of Grulla NWR is a former solid waste disposal site known as the Arch Landfill (Figure 2). This closed dump site encompasses 36 acres (15 hectares) and was in operation for over 50 years despite not being a state approved sanitary landfill (NMED, 1987; E&E, 1988). Wastes disposed of at this site included automotive, industrial, and oil field wastes, pesticide/herbicide containers, household garbage, cotton gin refuse, dead animals, and paint (NMED, 1987; E&E, 1988; NMED, 1989). In 1980, state, county, and local officials conducted a removal action at this site, transporting approximately 1,000 discarded pesticide containers from the dump to an approved landfill located in Portales, New Mexico (Fogg, 1980; NMED, 1980). At that time, the facility was designated as a modified landfill under

Figure 1: General Locations of Grulla National Wildlife Refuge and Muleshoe National Wildlife Refuge



county supervision, prohibited from receiving wastes other than household refuse (NMED, 1980; NMED, 1981). Over the next nine years, investigators from the State of New Mexico documented on several occasions that the site was receiving unauthorized wastes such as pesticide containers (NMED, 1981; NMED, 1987; NMED, 1988; NMED, 1989). Based on the results of these investigations in conjunction with the limited depth to groundwater in fill areas, the State of New Mexico recommended that the site be closed (NMED, 1989). The site ceased disposal operations in 1989 and was closed in accordance with state regulations (NMED, 1989). In 1988, prior to closure, a contractor representing the U.S. Environmental Protection Agency (USEPA) conducted limited soil and groundwater sampling at the site and in the arroyo that drains from the site into Grulla NWR, to address concerns raised by the State of New Mexico over possible contaminant migration from the site (NMED, 1987; E&E, 1988). Results of this limited sampling indicated that contamination at the dump did not appear to be migrating off site (E&E, 1988).

Located approximately 30 miles (48 kilometers) east of Grulla NWR is Muleshoe NWR (Figure 1). This refuge was established within the southern high plains in the Edwards Plateau Ecosystem in Bailey County, Texas, in 1935. The refuge is located approximately 20 miles (32 kilometers) south of the town of Muleshoe, Texas, and encompasses approximately 5,809 acres (2,351 hectares). Climate and rainfall are similar to Grulla NWR. Aquatic habitat within Muleshoe NWR consists of approximately 1,000 acres (405 hectares), divided among three saline lakes: White Lake (Upper and Lower), Goose Lake (Upper and Lower), and Paul's Lake (Upper and Lower). Soils surrounding the saline lakes are within the Drake and Potter series and similar to Grulla NWR, are generally alkaline (USDA, 1963). No known off-refuge sources discharge contaminants into the refuge, other than through global atmospheric deposition and/or stormwater run-off and possible unauthorized discharges from surrounding agricultural lands (primarily rangeland) and adjacent roadways.

MATERIALS & METHODS

Surficial grab soil samples were collected from six sites at Grulla NWR (Sites G01-G06) by USFWS personnel in August, 2003 (Table 1 and Figure 2). These sites were located in the dry arroyo down gradient of Arch Landfill and within the dry bed of Salt Lake. Additional surficial grab soil and/or sediment samples were collected from six sites at Muleshoe NWR for baseline purposes (Table 1 and Figure 3). Two of these sites were located in White Lake (Sites G07 and G08), while the remaining four sites were distributed evenly between Goose Lake (G09 and G10) and Paul's Lake (Sites G11 and G12). All six of the samples collected from Grulla NWR were classified as soils. Two of the six samples collected from Muleshoe NWR were characterized as sediments (from Sites G11 and G12 in Upper and Lower Paul's Lake), while the remaining four samples were considered soils. Muleshoe NWR was selected as the baseline area for this investigation because of its proximity to Grulla NWR, it has no known contaminant sources other than through global atmospheric deposition or from stormwater run-off and/or unauthorized discharges from surrounding rangelands and adjacent roadways, and both refuges have saline lake beds surrounded by alkaline soils.

Table 1. Location of sample sites at Grulla National Wildlife Refuge (G01-G06) and Muleshoe National Wildlife Refuge (G07-G12), 2003 (Note - latitude and longitude coordinates are in decimal degrees).

Sample Site	Latitude	Longitude	General Site Description
G01	34.078233	-103.09168	Dry lake bed at Grulla NWR.
G02	34.070872	-103.10960	Dry lake bed at Grulla NWR.
G03	34.069554	-103.11509	Dry lake bed at Grulla NWR.
G04	34.070528	-103.11882	Dry arroyo near confluence with Salt Lake at Grulla NWR.
G05	34.070057	-103.12347	Dry arroyo sloping from landfill towards Grulla NWR.
G06	34.071267	-103.12617	Dry arroyo immediately down gradient of landfill.
G07	33.946077	-102.77152	Dry bed of Upper White Lake at Muleshoe NWR.
G08	33.944611	-102.76951	Dry bed of Lower White Lake at Muleshoe NWR.
G09	33.962606	-102.74476	Dry bed of Lower Goose Lake at Muleshoe NWR.
G10	33.956822	-102.75345	Dry bed of Upper Goose Lake at Muleshoe NWR.
G11	33.983732	-102.71818	Sediment from Lower Paul's Lake near service road.
G12	33.982781	-102.71560	Sediment from Upper Paul's Lake near service road.

Each sample was collected at a depth of 0 to 6 inches [0 to 15 centimeters (cm)] using a disposable plastic scoop and placed in a pre-cleaned glass container. Once collected all samples were placed on ice in a cooler and transported to the USFWS Arlington, Texas ESFO. These samples remained refrigerated at 39 °F (4°C) until submitted through the Patuxent Analytical Control Facility (PACF) to contract laboratories for chemical analyses. Each sample was analyzed for moisture, sand, silt, and clay content, total metals (aluminum, arsenic, barium, beryllium, boron, cadmium, chromium, copper, iron, lead, magnesium, manganese, mercury, molybdenum, nickel, selenium, strontium, vanadium, and zinc), residual organochlorine pesticides [1,2,3,4-tetrachlorobenzene, 1,2,4,5-tetrachlorobenzene, aldrin, hexachlorobenzene (HCB), heptachlor, alpha hexachlorocyclohexane (α BHC), alpha (α) chlordane, beta hexachlorocyclohexane (β BHC), *cis*-nonachlor, delta hexachlorocyclohexane (δ BHC), dieldrin, endosulfan II, endrin, gamma hexachlorocyclohexane (γ BHC), gamma (γ) chlordane, heptachlor epoxide, mirex, *o,p'*-dichloro-diphenyl-dichloroethane (*o,p'*-DDD), *o,p'*-dichloro-diphenyl-dichloroethylene (*o,p'*-DDE), *o,p'*-dichloro-diphenyl-trichloroethane (*o,p'*-DDT), oxychlordane, *p,p'*-dichloro-diphenyl-dichloroethane (*p,p'*-DDD),

Figure 2: Sampling Points at Grulla National Wildlife Refuge, 2003.

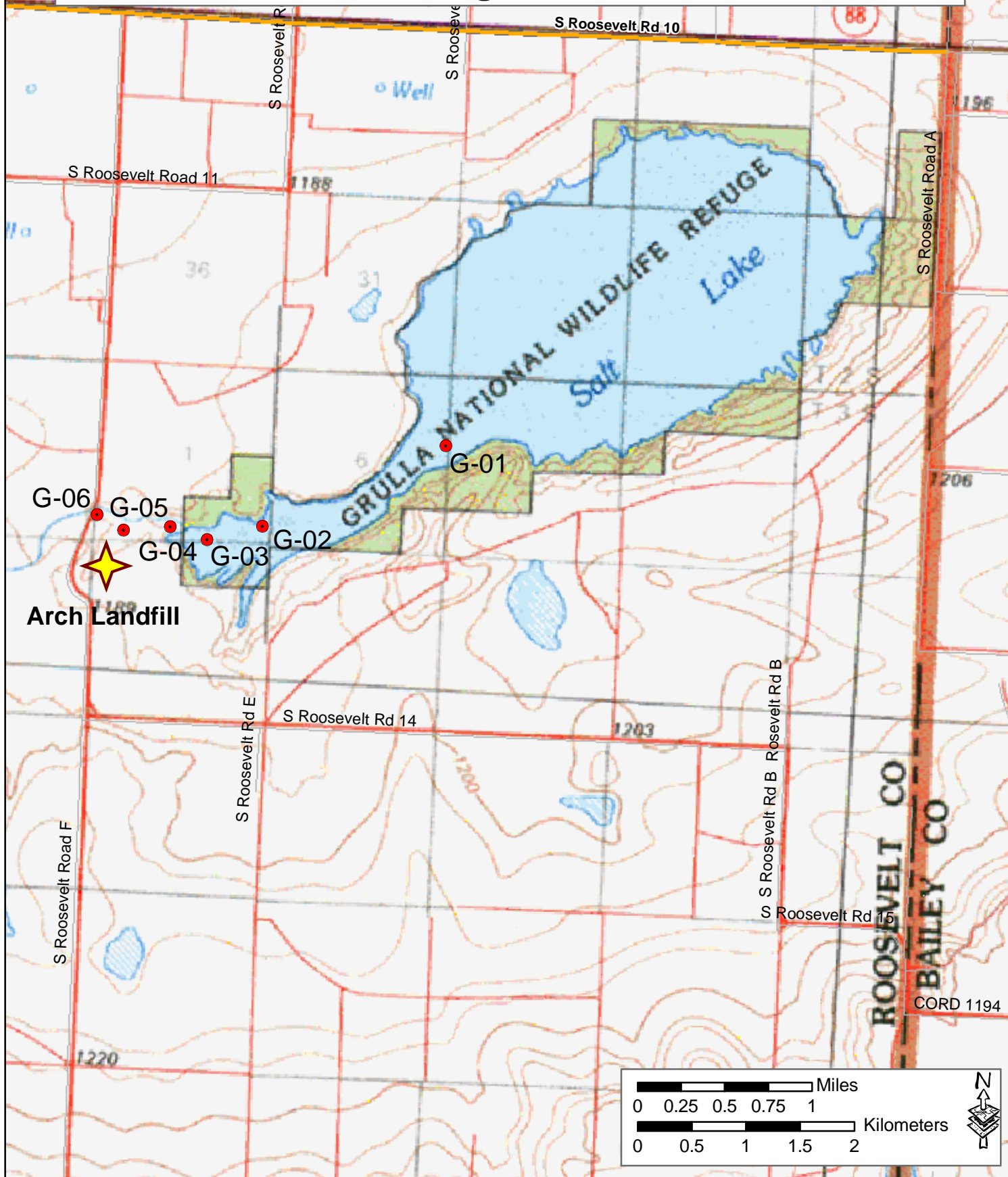
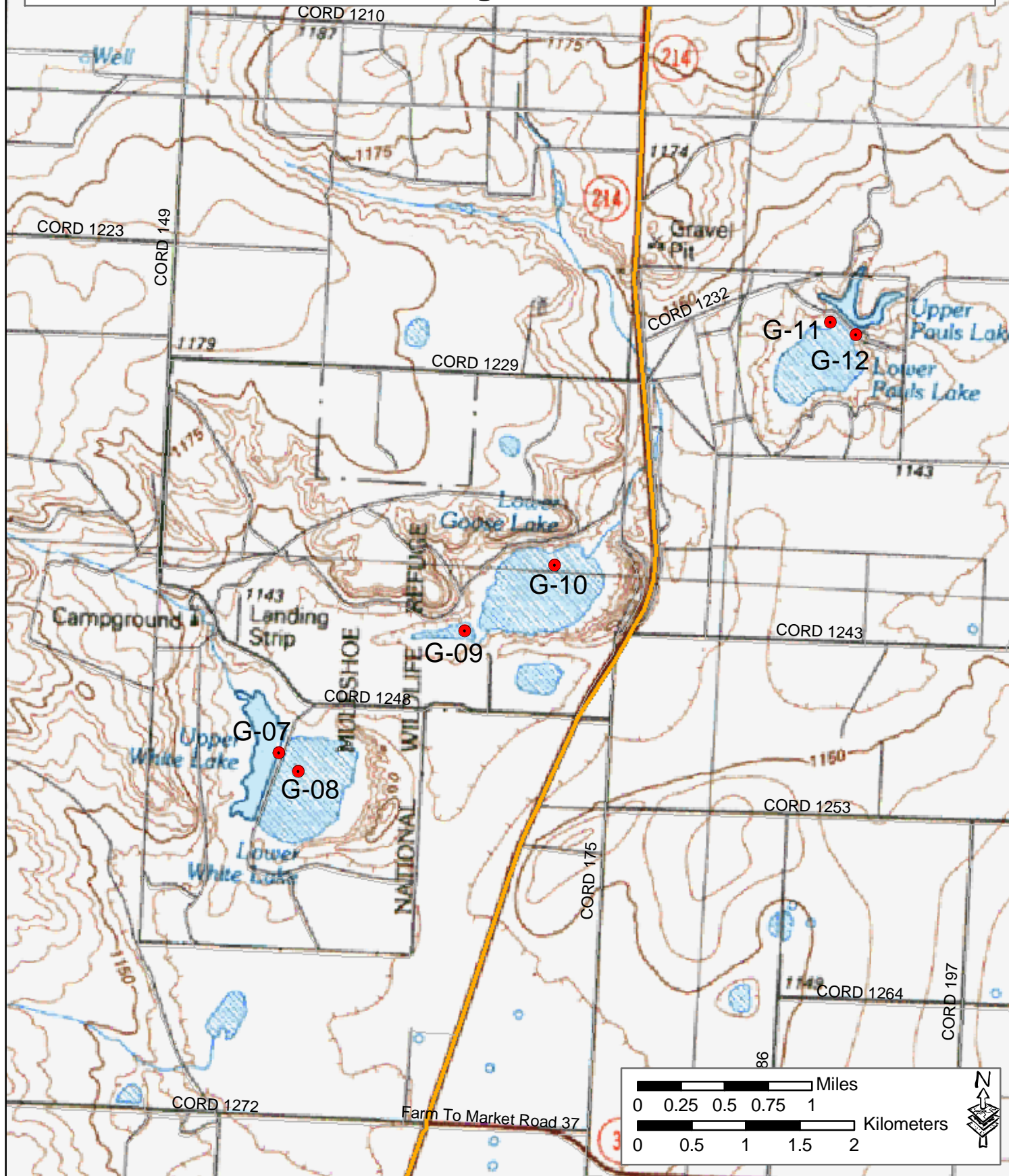


Figure 3: Sampling Points at Muleshoe National Wildlife Refuge, 2003.



p,p'-dichloro-diphenyl-dichloroethylene (*p,p'*-DDE), *p,p'*-dichloro-diphenyl-trichloroethane (*p,p'*-DDT), pentachloro-anisole, toxaphene, and *trans*-nonachlor], the organophosphate pesticide chlorpyrifos, total polychlorinated biphenyls (PCBs), and total petroleum hydrocarbons (TPH) (for analytical methods see Appendix A).

In addition to the soil and/or sediment sampling, another goal of this investigation was to sample the aquatic macroinvertebrate communities within lentic bodies at both Grulla NWR and Muleshoe NWR. The use of macroinvertebrates, especially insects, in the evaluation of water quality has been widely employed due in part to their abundance in a variety of aquatic habitats, ease of collection, sedentary nature, and an extensive range in response to environmental perturbations (Merritt and Cummins, 1996; Rosenberg and Resh, 1993). The data collected would be used as biotic indicators for assessing the ecological significance of measured contaminants between the two refuges. However, due to extended drought conditions at the time the samples were collected, Salt Lake and the arroyo draining from Arch Landfill into Grulla NWR were completely dry. White Lake, Goose Lake, and the majority of Lower Paul's Lake at Muleshoe NWR were also completely dry. Only a small, shallow pool area in the far northern portion of Lower Paul's Lake contained residual water. A sediment sample was collected from this area; however, there was not sufficient water to support aquatic life. Upper Paul's Lake at Muleshoe NWR was the only lentic body that contained sufficient water to allow for the collection of both macrobenthic and sediment samples. Macrobenthic samples were collected from Upper Paul's Lake using fine mesh dip nets and a Petite Ponar Grab following methodologies outlined by Kennedy *et al.* (1998). Once collected these samples were placed in polypropylene containers, preserved in 95% ethanol, and transported to the USFWS Arlington, Texas ESFO for identification to the lowest taxonomic level practical utilizing Thorp and Covich (1991) and Merritt and Cummins (1996).

RESULTS & DISCUSSION

The results of the chemical analyses are presented in Tables 2-7. The analytical results from Grulla NWR were compared to the results from Muleshoe NWR. In addition, where applicable, the analytical results from both refuges were compared with soil and/or sediment benchmarks, screening levels, and remedial target values proposed by the U.S. Environmental Protection Agency (USEPA), the Texas Commission on Environmental Quality (TCEQ), the New Mexico Environment Department (NMED), the Canadian Council of Ministers of the Environment (CCME), and the Ontario Ministry of the Environment (OME), as well as with other pertinent screening criteria such as background values, to assess the possible effects of contamination in soils and/or sediments collected from Grulla NWR and Muleshoe NWR.

Benchmarks and/or screening levels are values derived from toxicity data resulting from multiple studies. Soil benchmarks are typically based on the degree of toxicity of a given contaminant to plants, earthworms, heterotrophic microbes, and other invertebrates inhabiting soil regimes (Efroymson *et al.*, 1997). In contrast, remedial target values are soil cleanup levels usually

employed to address human health concerns. Sediment screening criteria have been developed based on the toxicological affects of a given contaminant to the biotic communities inhabiting benthic environments. For example, the OME considers the lowest effects level (LEL) indicative of a level of contamination that is non-toxic to the majority of benthic organisms, whereas the severe effect level (SEL) is indicative of contaminated sediments that would be detrimental to a majority of benthic organisms (Persaud *et al.*, 1993). In comparison, Long *et al.* (1995) state that the effects range-low (ER-L) of a detected chemical represents the lower 10th percentile of toxicological effects data for that specific chemical, whereas the effects range-median (ER-M) represents the toxicological effects data for the chemical at the 50th percentile. Concentrations detected below the ER-L represent a value where minimal effects would be expected, whereas concentrations detected at or above the ER-L but below the ER-M, represent a possible effects range (Long *et al.*, 1995). Concentrations detected at or above the ER-M represent a probable effects range where adverse toxicological effects would frequently occur (Long *et al.*, 1995). In a consensus based approach towards evaluating sediment screening criteria, Macdonald *et al.* (2000) report that the threshold effect concentration (TEC) for a contaminant in sediments is the concentration below which adverse effects are not expected, whereas the probable effect concentration (PEC) is the level above which adverse effects would likely occur. In the State of Texas, the TCEQ (2000) has developed 85th percentile values which are screening criteria for freshwater sediments based on the percentage of the lack of toxic effects to aquatic organisms from a given contaminant. As with soil benchmarks, LEL, SEL, ER-L, ER-M, TEC, PEC, and TCEQ 85th percentile values are non-regulatory screening guidelines developed to assist in assessing the degree of contamination in a given area.

Moisture, sand, silt, and clay content as percentages for the samples collected from six sites at Grulla NWR and six sites at Muleshoe NWR are presented in Table 2. Measured moisture content ranged

Table 2. Moisture, sand, silt, and clay content as percentages measured in soil/sediment samples collected from six sites at Grulla Lake National Wildlife Refuge (G01 - G06) and six sites at Muleshoe National Wildlife Refuge (G07 - G12), 2003.

Sample Site	% Moisture	% Sand	% Silt	% Clay
G01	22.8	27.1	60.2	12.6
G02	9.2	10.8	72.2	17.0
G03	20.3	55.6	42.2	2.2
G04	9.7	67.2	27.6	5.2
G05	7.0	78.7	14.4	6.9
G06	4.6	45.4	44.2	10.5
G07	32.5	8.0	74.8	17.1
G08	13.6	2.9	55.7	41.4
G09	26.0	4.7	78.4	16.8
G10	6.9	3.0	37.1	60.0
G11	30.2	23.0	36.1	40.9
G12	26.9	82.1	10.3	7.5

from 4.6% to 22.8% at Grulla NWR and from 6.9% to 32.5% at Muleshoe NWR. The soil samples collected from Sites G01 and G02 at Grulla NWR were dominated by silts, whereas the samples taken from Sites G03, G04, and G05 were predominantly composed of sands. The sample collected from Site G06 at Grulla NWR contained a homogeneous mixture of sands and silts. All of the samples collected at Muleshoe NWR were dominated by silts and clays, with the exception of the sediment sample collected from Site G12 which was composed primarily of sands (Table 2).

Metals

Results of the metals analyses for the six soil samples collected from Grulla NWR and the six soil/sediment samples collected from Muleshoe NWR are presented in Tables 3 and 4 in milligrams/kilogram (mg/kg) dry weight. Each sample was analyzed for 19 metallic constituents. Of these metals, mercury was the only analyte that was not detected above the analytical detection limits in any of the samples collected and is not considered further in this report.

[Aluminum (Al)] Approximately 8.1% of the Earth's crust is composed of aluminum (Miller and Gardiner, 1998). Background surface soil concentrations in the western U.S. can range up to 74,000 mg Al/kg (Shacklette and Boerngen, 1984). According to the TCEQ (2001), a soil-aluminum concentration of 30,000 mg Al/kg is considered background in the State of Texas. In wildlife, elevated levels of aluminum in the diet of birds can result in adverse effects in calcium and phosphorus metabolism (Sparling and Lowe, 1996). In an aqueous environment, the bio-availability of aluminum is driven by pH (Sparling and Lowe, 1996). Aluminum is relatively innocuous when the pH ranges from 5.5 to 7.5 but becomes soluble and biologically available when the pH is less than 5.5 (Sparling and Lowe, 1996). For many species of fish exposed to elevated levels of aluminum, toxic effects appear to correlate with decreasing pH, resulting in adverse effects that shift from asphyxiation to impaired ion regulation (Sparling and Lowe, 1996). According to Buchman (1999), the threshold effects level (TEL) for aluminum toxicity in freshwater sediments is 25,500 mg Al/kg dry weight.

Soil-aluminum levels at Grulla NWR ranged from 6,777 mg Al/kg dry weight at Site G05 to 15,847 mg Al/kg dry weight at Site G06 (Table 3), while soil-aluminum levels at Muleshoe NWR ranged from 11,938 mg Al/kg dry weight at Site G09 to 19,717 mg Al/kg dry weight at Site G10 (Table 4). The mean soil-aluminum concentration calculated for Grulla NWR [mean (\bar{x}) = 10,758.7 mg Al/kg dry weight; sample size (n) = 6] was less than the mean soil-aluminum concentration determined for Muleshoe NWR (\bar{x} = 15,494.3 mg Al/kg dry weight; n = 4). None of the detected soil-aluminum concentrations in any of the samples collected from either refuge equaled or exceeded the background values recommended by Shacklette and Boerngen (1984) and the TCEQ (2001).

The sediment-aluminum concentration measured at Site G11 (Lower Paul's Lake) was 8,170 mg Al/kg dry weight, while the sediment-aluminum level detected at Site G12 (Upper Paul's Lake) was 11,975 mg Al/kg dry weight. Both of these concentrations were well less than the sediment TEL suggested by Buchman (1999).

Table 3. Results of metals analysis in mg/kg dry weight for soil samples collected from six sites at Grulla Lake National Wildlife Refuge, Roosevelt County, New Mexico, 2003 (Note - dl is the analytical detection limit and bdl is below the analytical detection limit).

Analyte	G01	G02	G03	G04	G05	G06
Aluminum	11,242.00	14,031.00	9,692.00	6,963.00	6,777.00	15,847.00
dl	10.50	10.60	10.50	10.20	10.40	10.50
Arsenic	9.39	10.40	9.91	5.27	4.01	5.73
dl	0.53	0.53	0.52	0.51	0.52	0.53
Barium	173.00	232.00	206.00	203.00	257.00	264.00
dl	0.21	0.21	0.21	0.20	0.21	0.21
Beryllium	bdl	0.07	bdl	bdl	bdl	0.11
dl	0.02	0.02	0.02	0.02	0.02	0.02
Boron	73.20	82.70	45.00	18.70	22.00	17.60
dl	2.11	2.11	2.09	2.03	2.08	2.10
Cadmium	0.15	0.27	0.13	0.12	0.12	0.22
dl	0.11	0.11	0.11	0.10	0.10	0.11
Chromium	9.42	11.70	8.28	6.60	6.41	13.70
dl	0.53	0.53	0.52	0.51	0.52	0.53
Copper	4.69	10.20	5.56	5.93	4.23	9.24
dl	0.53	0.53	0.52	0.51	0.52	0.53
Iron	9,825.00	13,190.00	7,913.00	5,983.00	5,632.00	13,260.00
dl	10.50	10.60	10.50	10.20	10.40	10.50
Lead	4.00	10.90	4.77	4.30	3.86	10.60
dl	2.11	2.11	2.09	2.03	2.08	2.10
Magnesium	63,880.00	77,990.00	39,130.00	16,410.00	12,990.00	21,660.00
dl	10.50	10.60	10.50	10.20	10.40	10.50
Manganese	216.00	283.00	217.00	142.00	127.00	239.00
dl	1.05	1.06	1.05	1.02	1.04	1.05
Mercury	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.11	0.11	0.11	0.10	0.10	0.11
Molybdenum	3.71	4.02	1.28	bdl	bdl	bdl
dl	1.05	1.06	1.05	1.02	1.04	1.05
Nickel	8.33	11.20	6.87	5.94	5.24	11.60
dl	1.05	1.06	1.05	1.02	1.04	1.05
Selenium	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.53	0.53	0.52	0.51	0.52	0.53
Strontium	1,118.00	1,264.00	1,010.00	1,027.00	664.00	1,022.00
dl	0.53	0.53	0.52	0.51	0.52	0.53
Vanadium	42.10	63.10	44.40	32.70	26.30	47.00
dl	0.53	0.53	0.52	0.51	0.52	0.53
Zinc	24.70	31.10	20.10	20.10	15.50	35.20
dl	5.27	5.28	5.23	5.09	5.19	5.26

Table 4. Results of metals analysis in mg/kg dry weight for soil/sediment samples collected from six sites at Muleshoe Lake National Wildlife Refuge, Bailey County, Texas, 2003 (Note - dl is the analytical detection limit and bdl is below the analytical detection limit).

Analyte	G07	G08	G09	G10	G11	G12
Aluminum	14,850.00	15,472.00	11,938.00	19,717.00	8,170.00	11,975.00
dl	10.50	10.30	10.30	10.60	10.50	10.50
Arsenic	7.30	10.50	12.50	8.81	2.61	5.95
dl	0.52	0.51	0.51	0.53	0.52	0.52
Barium	197.00	291.00	269.00	332.00	99.40	160.00
dl	0.21	0.21	0.21	0.21	0.21	0.21
Beryllium	0.09	0.43	0.07	0.69	bdl	0.19
dl	0.02	0.02	0.02	0.02	0.02	0.02
Boron	134.00	45.60	65.80	17.70	78.80	34.30
dl	2.09	2.06	2.05	2.13	2.09	2.09
Cadmium	0.20	0.43	0.25	0.50	0.23	0.19
dl	0.10	0.10	0.10	0.11	0.11	0.10
Chromium	11.60	13.20	9.73	17.30	6.52	10.60
dl	0.52	0.51	0.51	0.53	0.52	0.52
Copper	8.93	10.90	9.08	18.10	6.66	9.42
dl	0.52	0.51	0.51	0.53	0.52	0.52
Iron	12,800.00	15,560.00	11,680.00	17,910.00	7,202.00	10,120.00
dl	10.50	10.30	10.30	10.60	10.50	10.50
Lead	8.40	10.80	8.56	19.90	6.51	9.39
dl	2.09	2.06	2.05	2.13	2.09	2.09
Magnesium	28,780.00	32,620.00	35,450.00	20,540.00	29,350.00	7,132.00
dl	10.50	10.30	10.30	10.60	10.50	10.50
Manganese	231.00	329.00	257.00	460.00	394.00	405.00
dl	1.05	1.03	1.03	1.06	1.05	1.05
Mercury	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.10	0.10	0.10	0.11	0.11	0.10
Molybdenum	3.81	bdl	4.13	bdl	5.21	bdl
dl	1.05	1.03	1.03	1.06	1.05	1.05
Nickel	10.60	14.70	10.70	19.60	6.95	12.30
dl	1.05	1.03	1.03	1.06	1.05	1.05
Selenium	bdl	bdl	bdl	bdl	bdl	2.12
dl	0.52	0.51	0.51	0.53	0.52	0.52
Strontium	1,214.00	1,678.00	1,935.00	984.00	642.00	501.00
dl	0.52	0.51	0.51	0.53	0.52	0.52
Vanadium	38.70	46.60	40.80	46.40	17.90	25.00
dl	0.52	0.51	0.51	0.53	0.52	0.52
Zinc	33.20	39.40	29.40	60.40	21.20	29.00
dl	5.23	5.14	5.13	5.32	5.23	5.23

[Arsenic (As)] According to Shacklette and Boerngen (1984), the estimated arithmetic mean for background elemental arsenic concentrations in surface soils in the western U.S. is 7 mg As/kg. The TCEQ (2001) considers a soil-arsenic concentration of 5.9 mg As/kg as background in the State of Texas. Pennington (1991) reported soil-arsenic concentrations ranging up to 13.4 mg As/kg in the Texas Panhandle. Efroymson *et al.* (1997) recommend an earthworm soils toxicity screening

benchmark of 60 mg As/kg dry weight, whereas the USEPA (2000) considers a soil-arsenic concentration of 37 mg As/kg dry weight as a benchmark value for terrestrial plants. In aquatic environments, elemental arsenic is insoluble in water, but many arsenic species are highly soluble in freshwater (Schneider, 1971). Common arsenic species include arsenate, arsenite, methanearsonic acid, and dimethyl arsenic acid (USEPA, 1980).

Toxic effects of arsenic to aquatic life are significantly dependent on numerous biological and abiotic factors, including water temperature, pH, organic content, phosphate concentrations, suspended solids, and arsenic speciation (Eisler, 1988a). In aerobic waters, reduced forms of arsenic tend to be oxidized into arsenates (USEPA, 1980). The adsorption of arsenate by metal oxides and the formation of arsenic sulfide appears to remove arsenic from the water column, binding it to the sediments, thereby preventing high concentrations of arsenic being present in solution (USEPA, 1980). The estimated residence time for arsenic in lentic systems is 45 years (Eisler, 1988a). The OME suggests a sediment LEL of 6 mg As/kg dry weight and a SEL of 33 mg As/kg dry weight (Persaud *et al.*, 1993), while Long *et al.* (1995), consider 8.2 mg As/kg dry weight as the ER-L for arsenic in sediments. MacDonald *et al.* (2000), recommend a sediment TEC of 9.79 mg As/kg dry weight and a PEC of 33 mg As/kg dry weight.

Soil-arsenic levels at Grulla NWR ranged from 4 mg As/kg dry weight at Site G05 to 10.4 mg As/kg dry weight at Site G02 (Table 3), while soil-arsenic levels at Muleshoe NWR ranged from 7.3 mg As/kg dry weight at Site G07 to 12.5 mg As/kg dry weight at Site G09 (Table 4). The mean soil-arsenic concentration determined for Grulla NWR (\bar{x} = 7.5 mg As/kg dry weight; n = 6) was less than the mean soil-arsenic concentration calculated for Muleshoe NWR (\bar{x} = 9.8 mg As/kg dry weight; n = 4). The detected soil-arsenic concentrations from Sites G01 (9.4 mg As/kg dry weight), G02, and G03 (9.9 mg As/kg dry weight) at Grulla NWR and in all soil samples collected from Muleshoe NWR exceeded the background values recommended by Shacklette and Boerngen (1984) and the TCEQ (2001). However, none of the soil samples collected from either refuge contained arsenic levels that exceeded the values reported by Pennington (1991), Efroymsen *et al.* (1997), or the USEPA (2000).

The sediment-arsenic concentration measured at Site G11 was 2.6 mg As/kg dry weight, while the sediment-arsenic level detected at Site G12 was 6 mg As/kg dry weight. The concentration measured at Site G12 equaled the lower toxicity threshold proposed by Persaud *et al.* (1993); however, neither of the two sites contained arsenic levels that exceeded the lower toxicity threshold values suggested for sediments by Long *et al.* (1995) and MacDonald *et al.* (2000).

[Barium (Ba)] Barium compounds are used in a variety of industrial applications. In nature, barium chiefly occurs as the relatively insoluble salts, barite and witherite (USEPA, 1986). Shacklette and Boerngen (1984) report an estimated arithmetic mean of 670 mg Ba/kg as background for soils in the western U.S., whereas a soils concentration of 300 mg Ba/kg dry weight is considered background in the State of Texas (TCEQ, 2001). According to Efroymsen *et al.* (1997), a concentration of 3,000 mg Ba/kg dry weight is the screening benchmark for barium toxicity to soil microorganisms, while the TCEQ (2001) considers a soil-barium concentration of 500 mg Ba/kg dry

weight as a benchmark value for terrestrial plants. The USEPA (2003a) reports concentrations of 330 mg Ba/kg dry weight and 1,000 mg Ba/kg dry weight as ecological soil screening levels for macroinvertebrate and mammalian wildlife receptors, respectively. In freshwater environments, the 85th percentile values reported by the TCEQ (2000) for sediment-barium levels range from 181 mg Ba/kg dry weight in lotic systems to 297 mg Ba/kg dry weight in lentic environments.

Soil-barium levels at Grulla NWR ranged from 173 mg Ba/kg dry weight at Site G01 to 264 mg Ba/kg dry weight at Site G06 (Table 3), while soil-barium levels at Muleshoe NWR ranged from 197 mg Ba/kg dry weight at Site G07 to 332 mg Ba/kg dry weight at Site G10 (Table 4). The mean soil-barium concentration calculated for Grulla NWR (\bar{x} = 222.5 mg Ba/kg dry weight; n = 6) was less than the mean soil-barium concentration calculated for Muleshoe NWR (\bar{x} = 272.3 mg Ba/kg dry weight; n = 4). The detected soil-barium concentrations in all of the samples collected from both refuges were less than the cited screening criteria for barium, with the exception of the sample collected from Site G10 at Muleshoe NWR. The concentration measured at this site slightly exceeded the TCEQ (2001) background value and the soil screening level for invertebrates recommended by the USEPA (2003a), but like all other samples collected, was less than the background value reported by Shacklette and Boerngen (1984) and well below the other ecological benchmarks suggested by Efroymson *et al.* (1997), the TCEQ (2001), and the USEPA (2003a).

The sediment-barium concentrations measured at Sites G11 and G12 were 99.4 mg Ba/kg dry weight and 160 mg Ba/kg dry weight, respectively. Both of these concentrations were less than the 85th percentile screening values reported by the TCEQ (2000) for sediments from lentic and lotic systems.

[Beryllium (Be)] Although not truly a heavy metal, beryllium is a rare element that is considered potentially toxic (Irwin and Dodson, 1991; Manahan, 1991). The distribution of beryllium in the environment largely results from the combustion of coal and oil (Goyer, 1991; Manahan, 1991). Coal mined from the mid-west U.S. contains an average of about 2.5 mg Be/kg while crude oil can contain approximately 0.08 mg Be/kg (Goyer, 1991). Beryllium concentrations in soils in the U.S. can range up to 15 mg Be/kg (Shacklette and Boerngen, 1984), but according to Shacklette and Boerngen (1984), the estimated arithmetic mean for background beryllium concentrations in soils in the western U.S. is 0.97 mg Be/kg. In the State of Texas, a soil-beryllium concentration of 1.5 mg Be/kg dry weight is considered background (TCEQ, 2001). The TCEQ (2001) recommends a soil-beryllium concentration of 10 mg Be/kg dry weight as a benchmark for terrestrial plants, whereas the USEPA (2003b) reports concentrations of 40 mg Be/kg dry weight and 36 mg Be/kg dry weight as ecological soil screening levels for macroinvertebrate and mammalian wildlife receptors, respectively. In freshwater environments, Irwin and Dodson (1991) state that in the absence of a known source, lotic systems usually contain very low or non-detectable amounts of beryllium. Currently, there are no screening criteria available to assess beryllium levels in sediments, but samples collected by the USFWS in 1996 from relatively undisturbed saline lakes in West Texas contained a mean concentration of 1.04 mg Be/kg dry weight (n = 4) (Irwin *et al.*, 1996).

Soil-beryllium concentrations were detected above the analytical detection limits in samples collected from only two sites at Grulla NWR, Sites G02 (0.07 mg Be/kg dry weight) and G06 (0.11 mg Be/kg dry weight) (Table 3). In contrast, all of the soil samples collected from Muleshoe NWR contained detectable amounts of beryllium (Table 4). The beryllium concentrations measured in soils taken from Muleshoe NWR ranged from 0.07 mg Be/kg dry weight at Site G09 to 0.69 mg Be/kg dry weight at Site G10 (Table 3). All of the soil-beryllium concentrations detected at both refuges were less than the background values reported by Shacklette and Boerngen (1984) and the TCEQ (2001) and well below the ecological benchmarks suggested by the TCEQ (2001) and the USEPA (2003).

No detectable amounts of beryllium were measured in the sediment sample collected from Site G11 (Table 4). The sediment-beryllium concentration detected in the sample collected from Site G12 equaled 0.19 mg Be/kg dry weight, well less than the mean value reported by Irwin *et al.* (1996) for beryllium levels measured in saline lakes from West Texas.

[Boron (B)] Boron compounds are used in the production of fertilizers and other agricultural chemicals such as herbicides and insecticides (Moore *et al.*, 1990; USDO, 1998). In the U.S., boron concentrations in soils typically range from 10-300 mg B/kg (USDO, 1998). Usually, arid, saline and/or alkaline soils will contain higher boron concentrations in comparison to watered, loamy soils (USDO, 1998). Furthermore, soils formed from marine sediments typically contain higher concentrations of boron than those formed from igneous rocks (Moore *et al.*, 1990). In saline soils, boron and other salts can occur naturally in toxic amounts (USDA, 1954). Xeric soils can contain up to 100 mg B/kg (RAIS, 2004a). According to Shacklette and Boerngen (1984), the estimated arithmetic mean for background boron concentrations in western soils is 29 mg B/kg, while in the State of Texas, a soils concentration of 30 mg B/kg is considered background (TCEQ, 2001). Efroymsen *et al.* (1997), recommend a screening benchmark value of 20 mg B/kg dry weight for boron toxicity to soil microorganisms and microbial processes, whereas the TCEQ (2001) considers a soil-boron concentration of 0.5 mg B/kg dry weight as a benchmark value for terrestrial plants. According to the U.S. Department of Agriculture (USDA) (1954), many plant species are susceptible to toxicity when soil-boron concentrations are greater than or equal to 1 mg B/kg. In aquatic systems, boron can react and bind with clays, suspended matter, and sediments (USDO, 1998). Eisler (1990) reports that freshwater sediments with a high clay composition usually contain less than 10 mg B/kg dry weight. Sediments collected by the USFWS in 1996 from four relatively undisturbed saline lakes in West Texas contained boron levels ranging from 4 mg B/kg dry weight to 60 mg B/kg dry weight (\bar{x} = 30 mg B/kg dry weight) (Irwin *et al.*, 1996).

Measured soil-boron levels at Grulla NWR ranged from 17.6 mg B/kg dry weight at Site G06 to 82.7 mg B/kg dry weight at Site G02 (Table 3), whereas soil-boron levels at Muleshoe NWR ranged from 17.7 mg B/kg dry weight at Site G10 to 134 mg B/kg dry weight at Site G07 (Table 4). The mean soil-boron concentration calculated for Grulla NWR (\bar{x} = 43.2 mg B/kg dry weight; n = 6) was less than the mean soil-boron concentration determined for Muleshoe NWR (\bar{x} = 65.8 mg B/kg dry weight; n = 4). The detected soil-boron concentrations from Sites G01 (73.2 mg B/kg dry weight), G02, and G03 (45 mg B/kg dry weight) at Grulla NWR and from Sites G07, G08 (45.6 mg B/kg dry

weight), and G09 (65.8 mg B/kg dry weight) at Muleshoe NWR exceeded the background values reported by Shacklette and Boerngen (1984) and the TCEQ (2001). In addition, all of the soil samples collected from both refuges contained boron levels that greatly exceeded the lower ecological thresholds suggested by the TCEQ (2001) and the USDA (1954), while the soil samples collected from Sites G01, G02, G03, and G05 (22 mg B/kg dry weight) at Grulla NWR and from Sites G07, G08, and G09 at Muleshoe NWR contained detectable amounts of boron that exceeded the upper ecological benchmark proposed by Efroymson *et al.* (1997). However, all of the soil samples collected from both refuges contained boron concentrations that fell within the background range reported by the U.S. Department of Interior (USDOI) (1998) and were less than the highest concentration reported for xeric soils (RAIS, 2004a). Consequently, the boron levels measured at these sites may be indicative of natural soil conditions rather than indicators of contamination associated with anthropogenic activities.

The sediment-boron concentration measured at Site G11 was 78.8 mg B/kg dry weight, while the sediment-boron level detected at Site G12 was 34.3 mg B/kg dry weight. Both of these sites contained sediment-boron levels that exceeded the value reported by Eisler (1990), but the concentration measured at Site G12 fell within the range reported by Irwin *et al.* (1996) for boron levels in sediments collected from saline lakes in West Texas. The boron concentration measured at Site G11 was elevated in comparison to the values reported by Irwin *et al.* (1996), but this may be attributed to the naturally high boron content associated with saline and/or alkaline soils present at Muleshoe NWR.

[Cadmium (Cd)] Ryan *et al.* (1980) reported that the normal range for elemental cadmium in surface soils in the U.S. is 0.06 to 0.5 mg Cd/kg. According to Efroymson *et al.* (1997), a proposed screening benchmark value for cadmium toxicity to soil microorganisms is 20 mg Cd/kg dry weight, while the TCEQ (2001) reports concentrations of 110 mg Cd/kg dry weight and 29 mg Cd/kg dry weight as ecological benchmarks for earthworms and terrestrial plants, respectively. The USEPA (2003c) reports ecological soil screening levels of 1 mg Cd/kg dry weight and 0.38 mg Cd/kg dry weight for avian and mammalian wildlife receptors, respectively. In aquatic systems, elemental cadmium is insoluble in water, whereas cadmium chloride, nitrate, and sulfate compounds are highly soluble in freshwater (Schneider, 1971). Cadmium toxicity in freshwater is moderated by increasing water hardness through either complexation with carbonate or competition with calcium ions (Wren *et al.*, 1995). In sediments, the OME recommends a LEL of 0.6 mg Cd/kg dry weight and a SEL of 10 mg Cd/kg dry weight (Persaud *et al.*, 1993), whereas Long *et al.* (1995), consider 1.2 mg Cd/kg dry weight as the ER-L for cadmium. MacDonald *et al.* (2000), suggest a sediment TEC of 0.99 mg Cd/kg dry weight and a PEC of 4.98 mg Cd/kg dry weight. Soil-cadmium levels at Grulla NWR ranged from 0.12 mg Cd/kg dry weight at Sites G04 and G05 to 0.27 mg Cd/kg dry weight at Site 2 (Table 3), whereas soil-cadmium levels at Muleshoe NWR ranged from 0.2 mg Cd/kg dry weight at Site G10 to 0.5 mg Cd/kg dry weight at Site G07 (Table 4). The mean soil-cadmium concentration calculated for Grulla NWR (\bar{x} = 0.17 mg Cd/kg dry weight; n = 6) was less than the mean soil-cadmium concentration calculated for Muleshoe NWR (\bar{x} = 0.35 mg Cd/kg dry weight; n = 4). The soil-cadmium concentrations measured in all of the samples collected from both refuges fell within the background range reported by Ryan *et al.* (1980) and all were well below the

ecological benchmarks suggested by Efroymson *et al.* (1997) and the TCEQ (2001). In addition, all of the samples contained cadmium concentrations that were less than the soil screening levels recommended by the USEPA (2003c), with the exception of the samples collected from Sites G07 and G08 (0.43 mg Cd/kg dry weight) at Muleshoe NWR. The concentrations measured at these two sites exceeded the cited screening level for mammals but were less than the screening criterion suggested for avian receptors (USEPA, 2003c).

Sediment-cadmium concentrations measured at Sites G11 and G12 were 0.23 mg Cd/kg dry weight and 0.19 mg Cd/kg dry weight, respectively. Both of these concentrations were less than the lower cadmium toxicity threshold concentrations recommended for sediments by Persaud *et al.* (1993), Long *et al.* (1995), and MacDonald *et al.* (2000).

[Chromium (Cr)] Shacklette and Boerngen (1984) reported an estimated arithmetic mean of 56 mg Cr/kg as background for soils in the western U.S. According to the TCEQ (2001), a soil-chromium concentration of 30 mg Cr/kg dry weight can be considered background in the State of Texas. Efroymson *et al.* (1997), proposed soil toxicity screening benchmark values ranging from 0.4 mg Cr/kg dry weight for earthworms to 10 mg Cr/kg dry weight for soil microorganisms. The USEPA (2000) considers a soil-chromium concentration of 5 mg Cr/kg dry weight as a benchmark value for terrestrial plants. In freshwater systems, hydrolysis and precipitation are more important physical processes in determining the fate of chromium in comparison to adsorption and bio-accumulation (Eisler, 1986). Chromium occurs in aqueous environments in various ionic forms, including the chromous, chromic, chromite, chromate, and/or dichromate ions (Becker and Thatcher, 1973). In the chromic or chromite forms, the ions are trivalent, whereas in the chromate and dichromate forms, the ions are hexavalent (Becker and Thatcher, 1973). Overall toxicity of chromium to aquatic biota is dependent on water hardness, temperature, pH, chemical speciation, and salinity, but in general, hexavalent chromium is more toxic than trivalent chromium (Becker and Thatcher, 1973; Eisler, 1986). According to Eisler (1986), the majority of chromium bound in sediments is unavailable for living organisms. The OME suggest a LEL of 26 mg Cr/kg dry weight and a SEL of 110 mg Cr/kg dry weight for chromium in sediments (Persaud *et al.*, 1993), whereas MacDonald *et al.* (2000), recommend a sediment TEC of 43.4 mg Cr/kg dry weight and a PEC of 111 mg Cr/kg dry weight.

Soil-chromium levels at Grulla NWR ranged from 6.4 mg Cr/kg dry weight at Sites G05 to 13.7 mg Cr/kg dry weight at Site G06 (Table 3), while soil-chromium levels at Muleshoe NWR ranged from 9.7 mg Cr/kg dry weight at Site G09 to 17.3 mg Cr/kg dry weight at Site G10 (Table 4). The mean soil-chromium concentration determined for Grulla NWR (\bar{x} = 9.35 mg Cr/kg dry weight; n = 6) was less than the mean soil-chromium concentration calculated for Muleshoe NWR (\bar{x} = 12.96 mg Cr/kg dry weight; n = 4). All of the soil samples collected from both refuges contained chromium levels that exceeded the lower ecological benchmarks suggested by Efroymson *et al.* (1997) and the USEPA (2000), while the samples collected from Sites G02 (11.7 mg Cr/kg dry weight) and G06 at Grulla NWR and from Sites G07 (11.6 mg Cr/kg dry weight), G08 (13.2 mg Cr/kg dry weight), and G10 at Muleshoe NWR contained chromium concentrations that exceeded the higher ecological benchmark recommended by the TCEQ (2001). However, none of the samples collected from either

refuge contained soil-chromium levels that were greater than the background values reported by Shacklette and Boerngen (1984) and the TCEQ (2001).

The sediment-chromium concentration measured at Site G11 was 6.5 mg Cr/kg dry weight, while the sediment-chromium level detected at Site G12 was 10.6 mg Cr/kg dry weight. Both of these concentrations were less than the lower chromium toxicity threshold levels suggested for sediments by Persaud *et al.* (1993) and MacDonald *et al.* (2000).

[Copper (Cu)] Copper is primarily used in the manufacturing of electrical equipment, pipe, and machinery (Eisler, 1998a). It is also an essential micronutrient that interacts in animals with other essential trace elements such as iron, zinc, molybdenum, manganese, nickel, and selenium and also with nonessential elements including silver, cadmium, mercury, and lead (Goyer, 1991; Eisler, 1998a). Enzymes associated with nitrate transformations in algae require copper (Horne and Goldman, 1994). In soils, Shacklette and Boerngen (1984), consider 27 mg Cu/kg as the arithmetic mean background copper concentration in the western U.S., while a soil-copper concentration of 15 mg/kg dry weight is considered background in the State of Texas (TCEQ, 2001). Efroymson *et al.* (1997) proposed a soils toxicity screening benchmark value of 100 mg Cu/kg dry weight. The TCEQ (2001) reports a value of 61 mg Cu/kg dry weight as the soil benchmark for earthworms. In aquatic environments, the type and amount of various copper compounds present in the water depends on water pH, temperature, alkalinity, and on the concentrations of bicarbonate, sulfide, and organic ligands (Eisler, 1998a). The solubility of copper and copper salts is decreased under reducing conditions and is further modified by pH, temperature, and hardness; size and density of suspended materials; rates of coagulation and sedimentation of particulates; and concentration of dissolved organics (Eisler, 1998a). Copper concentrations in sediment interstitial pore waters correlate positively with concentrations of dissolved copper in the overlying water column (Eisler, 1998a). Typically, sediment bound copper is available to benthic organisms under anoxic and low pH conditions (Eisler, 1998a). The OME recommends a sediment LEL of 16 mg Cu/kg dry weight and a SEL of 110 mg Cu/kg dry weight (Persaud *et al.*, 1993), whereas Long *et al.* (1995), consider 34 mg Cu/kg dry weight as the ER-L for copper in sediments. MacDonald *et al.* (2000), suggest a sediment TEC of 31.6 mg Cu/kg dry weight and a PEC of 149 mg Cu/kg dry weight.

Soil-copper levels at Grulla NWR ranged from 4.2 mg Cu/kg dry weight at Site G05 to 10.2 mg Cu/kg dry weight at Site G02 (Table 3), while soil-copper levels at Muleshoe NWR ranged from 8.9 mg Cu/kg dry weight at Site G07 to 18.1 mg Cu/kg dry weight at Site G10 (Table 4). The mean soil-copper concentration calculated for Grulla NWR (\bar{x} = 6.64 mg Cu/kg dry weight; n = 6) was less than the mean soil-copper concentration determined for Muleshoe NWR (\bar{x} = 11.75 mg Cu/kg dry weight; n = 4). All of the detected soil-copper concentrations in the samples collected from both refuges were less than the cited copper screening criteria, with the exception of the sample collected from Site G10 at Muleshoe NWR which contained a copper level that exceeded the background value reported by the TCEQ (2001), but was less than all other cited screening criteria (Shacklette and Boerngen, 1984; Efroymson *et al.*, 1997; TCEQ, 2001).

Sediment-copper concentrations measured at Sites G11 and G12 were 6.66 mg Cu/kg dry weight and 9.42 mg Cu/kg dry weight, respectively. Both of these concentrations were less than the lower copper toxicity threshold values recommended for sediments by Persaud *et al.* (1993), Long *et al.* (1995), and MacDonald *et al.* (2000).

[Iron (Fe)] Iron is a necessary nutrient that is a constituent of many enzymatic and other cellular processes (Horne and Goldman, 1994). It is absolutely essential both for the transport of oxygen to the tissues and for maintenance of oxidative systems within the tissue cells (Guyton, 1981). Iron composes approximately 5% of the Earth's crust (Miller and Gardiner, 1998). Background iron concentrations in surface soils in the western U.S. range up to 26,000 mg Fe/kg (Shacklette and Boerngen, 1984). In Texas, median background soil-iron concentrations are reported as 15,000 mg Fe/kg (TCEQ, 2001). Under normal oxidizing conditions in freshwater systems, ferric iron predominates over ferrous iron, and in turn, ferric iron forms insoluble compounds that rapidly disassociate from the water column and drop to the sediments (Horne and Goldman, 1994). The OME recommends a LEL of 20,000 mg Fe/kg dry weight and a SEL of 40,000 mg Fe/kg dry weight for iron in sediments (Persaud *et al.*, 1993). According to Beyer (1990), sediments from the Great Lakes containing less than 17,000 mg Fe/kg dry weight are considered non-polluted, whereas sediments containing iron concentrations greater than 25,000 mg Fe/kg dry weight are considered extremely polluted.

Soil-iron levels at Grulla NWR ranged from 5,632 mg Fe/kg dry weight at Site G05 to 13,260 mg Fe/kg dry weight at Site G06 (Table 3), while soil-iron levels at Muleshoe NWR ranged from 11,680 mg Fe/kg dry weight at Site G09 to 17,910 mg Fe/kg dry weight at Site G10 (Table 4). The mean soil-iron concentration determined for Grulla NWR (\bar{x} = 9,300.5 mg Fe/kg dry weight; n = 6) was less than the mean soil-iron concentration calculated for Muleshoe NWR (\bar{x} = 14,487.5 mg Fe/kg dry weight; n = 4). All of the samples collected from both refuges contained iron concentrations less than the cited background values, with the exception of the samples taken from Sites G08 and G10 at Muleshoe NWR, which contained iron levels (15,560 and 17,910 mg Fe/kg dry weight, respectively) that exceeded the background value reported by the TCEQ (2001), but were less than the concentration reported by Shacklette and Boerngen (1984).

The sediment-iron concentration measured at Site G11 was 7,202 mg Fe/kg dry weight, while the sediment-iron level detected at Site G12 was 10,120 mg Fe/kg dry weight. Both of these concentrations were less than the lower screening criterion indicative of iron contaminated sediments suggested by Persaud *et al.* (1993) and Beyer (1990).

[Lead (Pb)] Listed by the USEPA as a priority pollutant, lead is used in pigment and chemical production, metallurgy and steel manufacturing, storage batteries, ceramics, petroleum products, cable sheathing, pipe and sheeting fabrication, and ammunition production (Eisler, 1988b). Lead is neither essential nor beneficial to living organisms, and unlike mercury, lead does not exhibit bio-magnification through progressive trophic levels (Eisler, 1988b; Pain 1995). It is naturally occurring in soils and according to Shacklette and Boerngen (1984), the estimated arithmetic mean for background lead concentrations in surface soils in the western U.S. is 20 mg Pb/kg. The TCEQ

(2001), considers a soil-lead concentration of 15 mg Pb/kg dry weight as background in the State of Texas. The USEPA (2003d) reports ecological soil screening levels of 16 mg Pb/kg dry weight and 59 mg Pb/kg dry weight for avian and mammalian wildlife receptors, respectively. According to the TCEQ (2001) soil ecological screening criteria range from 50 mg Pb/kg dry weight for terrestrial plants to 500 mg Pb/kg dry weight for earthworms, whereas the USEPA (2003d) reports soil screening levels of 110 mg Pb/kg dry weight and 1,700 mg Pb/kg dry weight for the same receptor groups. In water, lead is most soluble and bio-available under conditions of low pH, low organic content, low concentrations of suspended sediments, and low concentrations of calcium, iron, manganese, zinc, and cadmium salts (Eisler, 1988b). The deposition of lead to sediments in aqueous environments is attributed primarily to the strong binding capacities of many sediment components for metals (Pain, 1995). Lead concentrations in aquatic plants have been directly correlated with sediment lead concentrations (Pain, 1995). The OME suggests a sediment LEL of 31 mg Pb/kg dry weight and a SEL of 250 mg Pb/kg dry weight (Persaud *et al.*, 1993), while Long *et al.* (1995), consider 47 mg Pb/kg dry weight as the ER-L for lead in sediments. MacDonald *et al.* (2000), suggest a sediment TEC of 35.8 mg Pb/kg dry weight and a PEC of 128 mg Pb/kg dry weight.

Soil-lead levels at Grulla NWR ranged from 3.9 mg Pb/kg dry weight at Site G05 to 10.9 mg Pb/kg dry weight at Site G02 (Table 3), while soil-lead levels at Muleshoe NWR ranged from 8.4 mg Pb/kg dry weight at Site G07 to 19.9 mg Pb/kg dry weight at Site G10 (Table 4). The mean soil-lead concentration calculated for Grulla NWR (\bar{x} = 6.41 mg Pb/kg dry weight; n = 6) was less than the mean soil-lead concentration determined for Muleshoe NWR (\bar{x} = 11.92 mg Pb/kg dry weight; n = 4). All of the detected soil-lead concentrations in the samples collected from both refuges were below the cited lead screening criteria, with the exception of the sample collected from Site G10 at Muleshoe NWR which contained a lead level that exceeded the background value reported by the TCEQ (2001) and the avian-soil screening level recommended by the USEPA (2003d), but was less than all other cited criteria (Shacklette and Boerngen, 1984; Efroymsen *et al.*, 1997; TCEQ, 2001; USEPA, 2003d).

The sediment-lead concentrations measured at Sites G11 and G12 were 6.51 mg Pb/kg dry weight and 9.39 mg Pb/kg dry weight, respectively. Both of these concentrations were well less than the lower toxicity threshold values recommended for sediments by Persaud *et al.* (1993), Long *et al.* (1995), and MacDonald *et al.* (2000).

[Magnesium (Mg)] Magnesium is an essential nutrient that is required for energy transfer in all living cells because it catalyzes the change from adenosine triphosphate (ATP) to adenosine diphosphate (ADP) (Horne and Goldman, 1994). The Earth's crust is composed of approximately 2.1% magnesium (Miller and Gardiner, 1998). Shacklette and Boerngen (1984), estimated the arithmetic mean for background magnesium concentrations in surface soils in the western U.S. as 10,000 mg Mg/kg. Saline or white alkali soils in the southwest U.S. typically contain high levels of magnesium, as well as elevated concentrations of sodium, calcium, and sometimes potassium (Brady, 1984; Lamond and Whitney, 1992). In freshwater environments, magnesium, along with calcium, is one of the two most common polyvalent metallic ions encountered (Cole, 1983; Irwin

and Dodson, 1991). Currently, there are no screening criteria available to assess the ecological significance of magnesium levels in sediments, but samples collected by the USFWS in 1996 from four relatively undisturbed saline lakes in West Texas contained magnesium concentrations ranging from 9,030 mg Mg/kg dry weight to 53,100 mg Mg/kg dry weight (\bar{x} = 22,683 mg Mg/kg dry weight) (Irwin *et al.*, 1996).

Soil-magnesium levels at Grulla NWR ranged from 12,990 mg Mg/kg dry weight at Site G05 to 77,990 mg Mg/kg dry weight at Site G02 (Table 3), while soil-magnesium levels at Muleshoe NWR ranged from 20,540 mg Mg/kg dry weight at Site G10 to 35,450 mg Mg/kg dry weight at Site G09 (Table 4). The mean soil-magnesium concentration determined for Grulla NWR (\bar{x} = 38,676.7 mg Mg/kg dry weight; n = 6) was greater than the mean soil-magnesium concentration calculated for Muleshoe NWR (\bar{x} = 29,347.5 mg Mg/kg dry weight; n = 4). All of the soil samples collected from both refuges contained magnesium concentrations that exceeded the background value reported by Shacklette and Boerngen (1984). This may be attributed to the naturally high magnesium content associated with saline soils present at both refuges.

The sediment-magnesium concentration measured at Site G11 was 29,350 mg Mg/kg dry weight, while the sediment-magnesium level detected at Site G12 was 7,132 mg Mg/kg dry weight. The magnesium concentrations measured at these two sites were not elevated in comparison to the range of values reported by Irwin *et al.* (1996) for saline lakes in West Texas.

[Manganese (Mn)] Manganese is a widely distributed, abundant element that constitutes approximately 0.085% of the earth's crust (Irwin and Dodson, 1991). It is a necessary nutrient for plants and animals that is relatively nontoxic to aquatic biota (Wiener and Giesy, 1979; Cole 1983). It stimulates planktonic growth in freshwater systems by activating enzymatic mechanisms (Cole, 1983). In surface soils, 480 mg Mn/kg is considered an estimated arithmetic mean background concentration in the western U.S. (Shacklette and Boerngen, 1984). The TCEQ (2001), considers a soil-manganese concentration of 300 mg Mn/kg dry weight as background in the State of Texas. According to Efroymson *et al.* (1997), a proposed screening benchmark value for manganese toxicity to soil microorganisms is 100 mg Mn/kg dry weight, whereas the TCEQ (2001) reports a soil-manganese concentration of 500 mg Mn/kg dry weight as a benchmark value for terrestrial plants. The ecological screening benchmark recommended by the USEPA for manganese in soils is 100 mg Mn/kg (RAIS, 2002). In sediments, the OME recommends a LEL of 460 mg Mn/kg dry weight and a SEL of 1,100 mg Mn/kg dry weight (Persaud *et al.*, 1993). In comparison, sediments from the Great Lakes containing less than 300 mg Mn/kg dry weight are considered non-polluted, whereas sediments containing manganese concentrations greater than 500 mg Mn/kg dry weight are considered heavily polluted (Beyer, 1990).

Soil-manganese levels at Grulla NWR ranged from 127 mg Mn/kg dry weight at Site G05 to 283 mg Mn/kg dry weight at Site G02 (Table 3), while soil-manganese levels at Muleshoe NWR ranged from 231 mg Mn/kg dry weight at Site G07 to 460 mg Mn/kg dry weight at Site G10 (Table 4). The mean soil-manganese concentration calculated for Grulla NWR (\bar{x} = 204 mg Mn/kg dry weight; n = 6) was less than the mean soil-manganese concentration calculated for Muleshoe NWR (\bar{x} = 319.3

mg Mn/kg dry weight; n = 4). The detected soil-manganese concentrations in all of the samples collected from both refuges were greater than the lower ecological threshold values proposed by Efroymson *et al.* (1997) and the USEPA (RAIS, 2002), while Sites G08 (329 mg Mn/kg dry weight) and G10 at Muleshoe NWR contained manganese concentrations that also exceeded the background value reported by the TCEQ (2001). However, none of the sites sampled at either refuge contained oil-manganese levels that exceeded the background value reported by Shacklette and Boerngen (1984) nor the upper ecological benchmark recommended by the TCEQ (2001).

Sediment-manganese concentrations measured at Sites G11 and G12 were 294 mg Mn/kg dry weight and 405 mg Mn/kg dry weight, respectively. Both of these concentrations were less than the lower manganese toxicity threshold value recommended for sediments by Persaud *et al.* (1993) and below the concentration indicative of a polluted system reported by Beyer (1990).

[Molybdenum (Mo)] Molybdenum is a comparatively rare element that does not occur free in nature and is usually found in conjunction with sulfur, oxygen, tungsten, lead, uranium, iron, magnesium, cobalt, vanadium, bismuth, or calcium (Eisler, 1989). It is an essential micronutrient for most life forms. It is even necessary for fixing atmospheric nitrogen by bacteria in plants; however, excessive exposure can result in toxicity to both animals and humans (Goyer, 1991; USDO, 1998). In terrestrial environments, the highest soil-molybdenum concentrations are usually found within the top 30 centimeters (12 inches) of surface soils (USDO, 1998). Ionic forms of molybdenum such as molybdate, tend to be sorbed most readily in alkaline soils which are high in calcium and chlorides, whereas retention is limited in low pH and low sulfate soils (Eisler, 1989). According to Shacklette and Boerngen (1984), the estimated arithmetic mean for background molybdenum concentrations in surface soils in the western U.S. is 1.1 mg Mo/kg. Efroymson *et al.* (1997), suggest a soils toxicity screening benchmark value of 200 mg Mo/kg dry weight for soil microorganisms, while the TCEQ (2001) considers a soils concentration of 2 mg Mo/kg as the benchmark value for terrestrial plants. Pastures containing between 20-100 mg Mo/kg may produce a disease in grazing animals known as teart (molybdenosis) which can prove fatal (Goyer, 1991). In freshwater systems with a pH greater than 7, molybdenum exists primarily as the molybdate ion, whereas at a pH less than 7, various polymeric compounds are formed, including the paramolybdate ion (Eisler, 1989). Aquatic organisms are relatively resistant to molybdenum (USDO, 1998). Background concentrations in sediments in the U.S. can range from 5 to 57 mg Mo/kg dry weight (USDO, 1998).

Soil-molybdenum concentrations were detected above the analytical detection limits in samples collected from only three sites at Grulla NWR, Sites G01 (3.71 mg Mo/kg dry weight), G02 (4.02 mg Mo/kg dry weight), and G03 (1.28 mg Mo/kg dry weight), and from only two sites at Muleshoe NWR, Sites G07 (3.81 mg Mo/kg dry weight) and G09 (4.13 mg Mo/kg dry weight) (Tables 3 and 4). All of these samples contained detectable amounts of molybdenum that exceeded the background value reported by Shacklette and Boerngen (1984), while the concentrations measured at Sites G01 and G02 at Grulla NWR and from Sites G07 and G09 at Muleshoe NWR also exceeded the lower ecological benchmark suggested by the TCEQ (2001); however, none of the sites at either refuge

contained molybdenum levels that approached the upper soil toxicity threshold proposed by Efroymson *et al.* (1997).

No detectable amounts of molybdenum were measured in the sediment sample collected from Site G12 (Table 3). The sediment-molybdenum concentration detected in the sample collected from Site G11 equaled 5.21 mg Mo/kg dry weight, well within the background range reported by the USDOI (1998) for sediments.

[Nickel (Ni)] Background surface soil-nickel concentrations can range up to 19 mg Ni/kg in the western U.S. and up to 10 mg Ni/kg in the State of Texas (Shacklette and Boerngen, 1984; TCEQ, 2001). According to Efroymson *et al.* (1997), a proposed screening benchmark value for nickel toxicity to soil microorganisms is 90 mg Ni/kg dry weight, while the TCEQ (2001) reports a soil-nickel concentration of 30 mg Ni/kg dry weight as a benchmark value for terrestrial plants. The physical and chemical forms of nickel and its salts strongly influence its bio-availability and toxicity in aqueous environments (Eisler, 1998b). In freshwater, nickel occurs as soluble salts adsorbed onto clay particles and organic matter (Eisler, 1998b). The distribution of nickel in an aquatic environment can be affected by pH, ionic strength, and availability of solid surfaces for adsorption (Eisler, 1998b). Sediment samples collected adjacent to a nickel smelter in Canada contained nickel concentrations as high as 5,000 mg Ni/kg dry weight, whereas sediments collected from lakes in the Rocky Mountains in the U.S. with no known sources other than background, contained nickel concentrations ranging from 10 to 18 mg Ni/kg dry weight (Eisler, 1998b). The OME recommends a sediment LEL of 16 mg Ni/kg dry weight and a SEL of 75 mg Ni/kg dry weight (Persaud *et al.*, 1993), whereas Long *et al.* (1995), recommend 21 mg Ni/kg dry weight as the ER-L for nickel in sediments. MacDonald *et al.* (2000), suggest a sediment TEC of 22.7 mg Ni/kg dry weight and a PEC of 48.6 mg Ni/kg dry weight.

Soil-nickel levels at Grulla NWR ranged from 5.2 mg Ni/kg dry weight at Site G05 to 11.6 mg Ni/kg dry weight at Site G06 (Table 3), while soil-nickel levels at Muleshoe NWR ranged from 10.6 mg Ni/kg dry weight at Site G07 to 19.6 mg Ni/kg dry weight at Site G10 (Table 4). The mean soil-nickel concentration calculated for Grulla NWR (\bar{x} = 8.2 mg Ni/kg dry weight; n = 6) was less than the mean soil-nickel concentration calculated for Muleshoe NWR (\bar{x} = 13.9 mg Ni/kg dry weight; n = 4). The detected soil-nickel concentrations in samples collected from two sites at Grulla NWR, Sites G02 (11.2 mg Ni/kg dry weight) and G06, and from all of the sites at Muleshoe NWR exceeded the background value reported by the TCEQ (2001), while the sample from Site G10 at Muleshoe NWR also exceeded the background concentration reported by Shacklette and Boerngen (1984). However, none of the samples collected from either refuge contained soil-nickel levels that exceeded the ecological benchmarks suggested by Efroymson *et al.* (1997) or the TCEQ (2001).

The sediment-nickel concentration measured at Site G11 was 6.95 mg Ni/kg dry weight, while the sediment-nickel level detected at Site G12 was 12.3 mg Ni/kg dry weight. The nickel concentrations measured at both of these sites were less than the lower nickel toxicity threshold values recommended for sediments by Persaud *et al.* (1993), Long *et al.* (1995), and MacDonald *et al.* (2000).

[Selenium (Se)] Selenium is an essential micronutrient but like other necessary dietary minerals, elevated levels can have detrimental effects on exposed organisms. It typically exists in nature and biotic systems as either selenate, selenite, elemental selenium, and/or selenide (Eisler, 1985; Goyer, 1991). Selenium volatilizes from soils and sediments at rates that are modified by temperature, moisture, time, season of year, concentration of water soluble selenium, and microbial activity (Eisler, 1985). According to Shacklette and Boerngen (1984), the estimated arithmetic mean for background selenium concentrations in surface soils in the western U.S. is 0.34 mg Se/kg. In Texas, a soil-selenium concentration of 0.3 mg Se/kg dry weight is considered background (TCEQ, 2001). The TCEQ (2001) reports soil-selenium concentrations of 1 mg Se/kg dry weight and 70 mg Se/kg as benchmark values for plants and earthworms, respectively. In an aqueous environment, selenium concentrations are a function of selenium levels contained within the drainage system and water pH (Eisler, 1985). In sediments, elemental selenium has a tendency to predominate in reducing environments (Van Derveer and Canton, 1997). According to Van Derveer and Canton (1997), the predicted effects concentration of selenium in sediments would be 2.5 mg Se/kg, while the observed effects threshold for fish and wildlife toxicity would be 4 mg Se/kg.

None of the soil samples collected from either refuge nor the sediment sample taken from Site G11 at Muleshoe NWR contained detectable amounts of selenium (Tables 3 and 4). The sediment-selenium concentration detected at Site G12 at Muleshoe NWR equaled 2.12 mg Se/kg dry weight, below the toxicity threshold values recommended for sediments by Van Derveer and Canton (1997).

[Strontium (Sr)] Strontium is a fairly common alkaline earth metal that is used in the manufacturing of pyrotechnics including signal flares and tracer bullets, the production of glass and ceramics, and sugar refining (Merck, 1989; Irwin and Dodson, 1991). Irwin *et al.* (1997) report that typical surficial soils contain strontium levels ranging anywhere from 5 mg Sr/kg to 3,000 mg Sr/kg, while arid, desert soils can contain strontium levels ranging up to 2,000 mg Sr/kg (RAIS, 2004a). According to Shacklette and Boerngen (1984), the estimated arithmetic mean for background strontium concentrations in western soils in the U.S. is 270 mg Sr/kg, whereas a soils concentration of 100 mg Sr/kg is considered background in the State of Texas (TCEQ, 2001). In localities where it is abundant, strontium like calcium, is an important freshwater quality ion that contributes to water hardness (Irwin and Dodson, 1991). Buchman (1999) considers 49 mg Sr/kg dry weight to be the background level for strontium in freshwater sediments, whereas sediments collected by the USFWS in 1996 from four relatively undisturbed saline lakes in West Texas contained a mean strontium concentration of 1,200 mg Sr/kg dry weight (Irwin *et al.*, 1996).

Soil-strontium levels at Grulla NWR ranged from 664 mg Sr/kg dry weight at Site G05 to 1,264 mg Sr/kg dry weight at Site G02 (Table 3), while soil-strontium levels at Muleshoe NWR ranged from 984 mg Sr/kg dry weight at Site G10 to 1,935 mg Sr/kg dry weight at Site G09 (Table 4). The mean soil-strontium concentration calculated for Grulla NWR (\bar{x} = 1,017.5 mg Sr/kg dry weight; n = 6) was less than the mean soil-strontium concentration calculated for Muleshoe NWR (\bar{x} = 1,452.8 mg Sr/kg dry weight; n = 4). All of the soil samples collected from both refuges contained strontium levels that exceeded the background concentrations suggested by Shacklette and Boerngen (1984) and the TCEQ (2001), but fell within the range reported by Irwin *et al.* (1997).

Sediment-strontium concentrations measured at Sites G11 and G12 were 501 mg Sr/kg dry weight and 642 mg Sr/kg dry weight, respectively. Both of these concentrations exceeded the sediment background concentration suggested by Buchman (1999), but both sites contained strontium levels less than the mean sediment concentration reported by Irwin *et al.* (1996) for saline lakes in West Texas.

[Vanadium (V)] Approximately 0.01% of the Earth's crust is composed of vanadium (Merck, 1989). Vanadium compounds are used in the production of rust-resistant metals, the manufacturing of ammunition, in x-rays, as catalysts in the distillation of alcohols and the production of synthetic rubber, and to reduce mercuric and ferric salts to mercurous and ferrous salts in industrial processes (Sax and Lewis, 1987; Merck, 1989). Vanadium is also a trace component of fossil fuels (Merck, 1989; ETC, 2000). Crude oil from West Texas contains approximately 3.2 mg V/Liter (ETC, 2000). In soils, vanadium concentrations can range up to 500 mg V/kg in the U.S. (Shacklette and Boerngen, 1984). The estimated arithmetic mean for background vanadium concentrations in soils in the western U.S. is 88 mg V/kg (Shacklette and Boerngen, 1984), while a soils concentration of 50 mg V/kg dry weight is considered background in the State of Texas (TCEQ, 2001). The ecological benchmark recommended by the USEPA for vanadium in soil is 2 mg V/kg (RAIS, 2002). Efroymsen *et al.* (1997), proposed a screening criterion of 20 mg V/kg for soil microorganisms, while the TCEQ (2001) considers a soil-vanadium concentration of 2 mg V/kg dry weight as a benchmark value for terrestrial plants. In freshwater systems, Buchman (1999) considers a vanadium concentration of 50 mg V/kg dry weight as the background value for sediments.

Soil-vanadium levels at Grulla NWR ranged from 26.3 mg V/kg dry weight at Site G05 to 63.1 mg V/kg dry weight at Site G02 (Table 3), while soil-vanadium levels at Muleshoe NWR ranged from 38.7 mg V/kg dry weight at Site G07 to 46.6 mg V/kg dry weight at Site G08 (Table 4). The mean soil-vanadium concentration calculated for Grulla NWR (\bar{x} = 42.6 mg V/kg dry weight; n = 6) was less than the mean soil-vanadium concentration determined for Muleshoe NWR (\bar{x} = 43.1 mg V/kg dry weight; n = 4). All of the detected soil-vanadium concentrations in the samples collected from both refuges exceeded the ecological benchmarks suggested by Efroymsen *et al.* (1997) and the TCEQ (2001). The sample collected from Site G02 at Grulla NWR also contained a vanadium level that exceeded the background value reported by the TCEQ (2001), but none of the soil samples taken from either refuge contained vanadium concentrations that were greater than the background value reported by Shacklette and Boerngen (1984).

Sediment-vanadium concentrations measured at Sites G11 and G12 were 17.9 mg V/kg dry weight and 25 mg V/kg dry weight, respectively. Both of these concentrations were less than the sediment screening criterion reported by Buchman (1999).

[Zinc (Zn)] Zinc is a naturally occurring metallic element found in soil but is also listed by the USEPA as a priority pollutant. It is used in the production of non-corrosive alloys and brass and in galvanizing steel and iron products (Eisler, 1993). Shacklette and Boerngen (1984), estimated the arithmetic mean for background zinc concentrations in surface soils in the western U.S. at 65 mg

Zn/kg. The TCEQ (2001), considers a soil-zinc concentration of 30 mg Zn/kg as background in the State of Texas. Efroymson *et al.* (1997), proposed a soils toxicity screening benchmark value of 100 mg Zn/kg dry weight for soil microorganisms and invertebrates, whereas the ecological screening benchmark recommended by the USEPA for zinc in soils is 120 mg Zn/kg (RAIS, 2003).

According to Eisler (1993), the majority of zinc introduced into an aquatic environment is partitioned into the sediment. Bio-availability of zinc from sediments is enhanced under conditions of high dissolved oxygen, low salinity, low pH, and high levels of inorganic oxides and humic substances (Eisler, 1993). Sediment-zinc concentrations less than 90 mg Zn/kg dry weight are considered supportive of aquatic biota, whereas zinc concentrations greater than 200 mg Zn/kg dry weight can be harmful to aquatic biota (Eisler, 1993). The OME recommends a sediment LEL of 120 mg Zn/kg dry weight and a SEL of 820 mg Zn/kg dry weight (Persaud *et al.*, 1993), while Long *et al.* (1995), consider 150 mg Zn/kg dry weight as the ER-L for zinc in sediments. MacDonald *et al.* (2000), suggest a sediment TEC of 121 mg Zn/kg dry weight and a PEC of 459 mg Zn/kg dry weight.

Soil-zinc levels at Grulla NWR ranged from 15.5 mg Zn/kg dry weight at Site G05 to 35 mg Zn/kg dry weight at Site G06 (Table 3), while soil-zinc levels at Muleshoe NWR ranged from 29.4 mg Zn/kg dry weight at Site G09 to 60.4 mg Ni/kg dry weight at Site G10 (Table 4). The mean soil-zinc concentration calculated for Grulla NWR (\bar{x} = 24.5 mg Zn/kg dry weight; n = 6) was less than the mean soil-zinc concentration calculated for Muleshoe NWR (\bar{x} = 40.6 mg Zn/kg dry weight; n = 4). The detected soil-zinc concentrations in samples collected from two sites at Grulla NWR, Sites G02 (31.1 mg Zn/kg dry weight) and G06, and from three sites at Muleshoe NWR, Sites G07 (33.2 mg Zn/kg dry weight), G08 (39.4 mg Zn/kg dry weight), and G10, exceeded the background value reported by the TCEQ (2001). However, none of the soil samples collected from either refuge contained zinc levels that exceeded the background concentration reported by Shacklette and Boerngen (1984) or the ecological benchmarks suggested by Efroymson *et al.* (1997) and the USEPA (RAIS, 2003).

The sediment-zinc concentration measured at Site G11 was 21.2 mg Zn/kg dry weight, while the sediment-zinc level detected at Site G12 was 29 mg Zn/kg dry weight. The zinc concentrations measured at both of these sites were less than the lower toxicity threshold values recommended for sediments by Persaud *et al.* (1993), Long *et al.* (1995), and MacDonald *et al.* (2000).

Organochlorine Pesticides

Results of the organochlorine pesticide analyses for the six soil samples collected from Grulla NWR and the six soil/sediment samples collected from Muleshoe NWR are presented in Tables 5 and 6 in mg/kg dry weight. Each sample was analyzed for 28 compounds. Of these compounds, only six [1,2,4,5-tetrachlorobenzene, *cis*-nonachlor, oxychlordan, *o,p'*-dichloro-diphenyl-trichloroethane (*o,p'*-DDT), *p,p'*-dichloro-diphenyl-dichloroethylene (*p,p'*-DDE), and *p,p'*-dichloro-diphenyl-trichloroethane (*p,p'*-DDT)] were detected above the analytical detection limits in any of the samples collected.

Table 5. Results of organochlorine pesticide analyses in mg/kg dry weight for six soil samples collected from Grulla National Wildlife Refuge, Roosevelt County, New Mexico, 2003 (Note- dl is the analytical detection limit; bdl is below the analytical detection limit; and ★ is detected above the analytical detection limit).

Analyte	G01	G02	G03	G04	G05	G06
1,2,3,4-tetrachlorobenzene	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
1,2,4,5-tetrachlorobenzene ★	0.000856	0.000722	bdl	0.000720	bdl	0.000614
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
aldrin	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
HCB	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
heptachlor	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
αBHC	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
αchlordane	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
βBHC	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
cis-nonachlor	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
δBHC	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
dieldrin	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
endosulfan II	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
endrin	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
γBHC	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
γchlordane	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
heptachlor epoxide	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
mirex	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
o,p'-DDD	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
o,p'-DDE	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
o,p'-DDT	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
oxychlordane ★	bdl	bdl	bdl	bdl	0.001080	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
p,p'-DDD	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
p,p'-DDE ★	bdl	bdl	bdl	bdl	bdl	0.001100
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
p,p'-DDT ★	bdl	bdl	bdl	bdl	bdl	0.000658
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
pentachloro-anisole	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
toxaphene	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.001570	0.001360	0.001560	0.001370	0.001340	0.001290
trans-nonachlor	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514
chlorpyrifos	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000626	0.000543	0.000623	0.000547	0.000537	0.000514

Table 6. Results of organochlorine pesticide analyses in mg/kg dry weight for six soil/sediment samples collected from Muleshoe National Wildlife Refuge, Bailey County, Texas, 2003 (Note - dl is the analytical detection limit; bdl is below the analytical detection limit; and ★ is detected above the analytical detection limit).

Analyte	G07	G08	G09	G10	G11	G12
1,2,3,4-tetrachlorobenzene	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
1,2,4,5-tetrachlorobenzene ★	0.001120	0.000915	0.000872	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
aldrin	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
HCB	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
heptachlor	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
αBHC	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
αchlordane	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
βBHC	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
cis-nonachlor ★	bdl	bdl	bdl	bdl	0.002550	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
δBHC	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
dieldrin	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
endosulfan II	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
endrin	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
γBHC	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
γchlordane	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
heptachlor epoxide	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
mirex	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
o,p'-DDD	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
o,p'-DDE	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
o,p'-DDT ★	bdl	bdl	bdl	bdl	0.000835	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
oxychlordane	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
p,p'-DDD	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
p,p'-DDE ★	0.001870	bdl	bdl	0.007300	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
p,p'-DDT	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
pentachloro-anisole	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
toxaphene	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.001830	0.001440	0.001670	0.001330	0.001780	0.001680
trans-nonachlor	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671
chlorpyrifos	bdl	bdl	bdl	bdl	bdl	bdl
dl	0.000732	0.000577	0.000670	0.000532	0.000713	0.000671

[1,2,4,5-Tetrachlorobenzene] Listed by the USEPA as a persistent, bio-accumulative, and toxic chemical (PBT), 1,2,4,5-tetrachlorobenzene is a common component of many herbicides, insecticides, defoliants, and electrical insulation fluids (Sax and Lewis, 1987; NDDH, 2002). In soils, the USEPA considers a concentration of 0.01 mg/kg as an ecological benchmark value (RAIS,

2003). Buchman (1999) suggests a remedial target value of 0.1 mg/kg for residual chlorobenzenes as a group, while the remedial level for residential soils in Texas is 1.1 mg/kg (TAC, 1993). The NMED (2000) reports a soil screening concentration of 0.16 mg/kg to be protective of construction workers involved in earth moving activities. In aquatic environments, the 85th percentile screening criterion reported by the TCEQ (2000) for sediments from freshwater lotic systems is 0.67 mg/kg dry weight, whereas the USEPA considers a concentration of 20.9 mg/kg as a sediment benchmark (RAIS, 2003).

This compound was detected above the analytical detection limits in soil samples collected from four sites at Grulla NWR (G01, G02, G04, and G06) and from three sites at Muleshoe NWR (G07, G08, and G09) (Tables 5 and 6). None of the sediment samples collected from Muleshoe NWR contained detectable amounts of 1,2,4,5-tetrachlorobenzene (Table 6). The soil concentrations detected at Grulla NWR ranged from 0.00061 mg/kg dry weight at Site G06 to 0.00086 mg/kg dry weight at Site G01 (Table 5). The soil concentrations measured at Muleshoe NWR ranged from 0.00087 mg/kg dry weight at Site G09 to 0.00112 mg/kg dry weight at Site G07 (Table 6). The arithmetic mean concentration calculated for the samples collected from Grulla NWR (\bar{x} = 0.00073 mg/kg dry weight; n = 4) was less than the mean concentration determined for Muleshoe NWR (\bar{x} = 0.00097 mg/kg dry weight; n = 3). The detected 1,2,4,5-tetrachlorobenzene concentrations at both refuges were below all cited soil screening criteria (TAC, 1993; Buchman, 1999; NMED, 2000; RAIS, 2003).

[Chlordane, isomers, and metabolites)] Technical chlordane consists of the stereoisomers alpha (α) and gamma (γ) or *cis* and *trans*-chlordane, heptachlor, *cis*- and *trans*-nonachlor, and the metabolites oxychlordane and heptachlor epoxide (ATSDR, 1994). First developed in 1946, chlordane was used as a general pesticide until 1983 (LMF, 2002). Between 1983 and 1988, use of chlordane in the United States was restricted by the USEPA to subterranean termite control (ATSDR, 1994). All commercial use of chlordane as a pesticide was banned by the USEPA in the United States in 1988 (ATSDR, 1994).

Once in the environment, chlordane binds tightly with soil particles and can remain in soils for more than 20 years (LMF, 2002). It can bio-accumulate in the tissues of fish, birds, and mammals and can adversely affect the nervous, digestive, and hepatic systems in both humans and animals (ATSDR, 1994; LMF, 2002). In soils, the USEPA considers a chlordane concentration of 0.224 mg/kg as an ecological benchmark value (RAIS, 2003), whereas the screening criterion for coarse textured agricultural, residential, and parkland soils in Ontario, Canada, is 0.29 mg/kg (EPT, 1999). The TCEQ recommends a soil-chlordane concentration of 0.49 mg/kg as protective of human health in residential areas in Texas (TAC, 1993), while in New Mexico, the NMED (2000) considers a soil concentration of 0.11 mg/kg to be protective of construction workers involved in earth moving activities. In aquatic systems, the OME reports a no effect level of 0.005 mg/kg dry weight for technical chlordane in sediments and recommends a LEL of 0.007 mg/kg dry weight and a SEL of 0.06 mg/kg dry weight (Persaud *et al.*, 1993). The TCEQ (2001) considers a concentration of 0.0045 mg/kg dry weight as an ecological benchmark for sediments, while MacDonald *et al.* (2000), suggest a sediment TEC of 0.0032 mg/kg dry weight and a PEC of 0.018 mg/kg dry weight.

The chlordane metabolite, oxychlordane, was detected above the analytical detection limit in soil collected from one site at Grulla NWR (Site G05), while the isomer *cis*-nonachlor was measured above the analytical detection limit in sediment collected from one site at Muleshoe NWR (Site G11) (Tables 5 and 6). No other chlordane isomers and/or metabolites were detected above the analytical detection limits in any of the remaining samples collected. The metabolite concentration measured at Site G05 (0.0011 mg/kg dry weight) was less than all cited soil-chlordane screening criteria (TAC, 1993; EPT, 1999; RAIS, 2003). The *cis*-nonachlor level detected at Site G11 (0.0026 mg/kg dry weight) was below all cited sediment screening criteria for technical chlordane (Persaud *et al.*, 1993; MacDonald *et al.*, 2000; TCEQ, 2001).

[Dichloro-diphenyl-trichloroethane (DDT), isomers, and metabolites] First developed in 1939, dichloro-diphenyl-trichloroethane (DDT) was used extensively throughout the world as an insecticide (ATSDR, 1995). Considered a probable human carcinogen by the USEPA, commercial production of DDT was banned in the United States in 1972 because of adverse affects to non-target wildlife species and the potential harm to human health (ATSDR, 1995; ATSDR, 2000a). The metabolites dichloro-diphenyl-dichloroethane (DDD) and dichloro-diphenyl-dichloroethylene (DDE) are microbial degradation products formed by the dehydrohalogenation of DDT (ATSDR, 2000a). In wildlife, DDT exposure has resulted in birds, alligators, and turtles producing eggs with shells too thin for offspring survival (Baskin, 2002). This compound exhibits very low solubility in aquatic environments and bio-accumulates in the fatty tissues of fish, birds, and other animals (Baskin, 2002). In soils, DDT binds readily to soil particles, with a half life estimated at 2 to 15 years (ATSDR, 1995). In Canada, the screening criteria for DDD, DDE, and DDT in coarse textured agricultural, residential, and parkland soils are 2.2, 1.6, and 1.6 mg/kg, respectively (EPT, 1999). The TCEQ considers soil-DDD, -DDE, and -DDT concentrations of 2.7, 1.9, and 1.9 mg/kg, respectively, as remedial target values in residential areas in Texas (TAC, 1993), while in New Mexico, the NMED (2000) reports a level of 0.27 mg/kg for DDD, DDE, and DDT to be protective of construction workers involved in earth moving activities. For total-DDT (the sum of all isomers and metabolites), the USEPA recommends a soil screening criterion of 0.0025 mg/kg (RAIS, 2002), while the CCME recommends a screening criterion of 0.7 mg/kg for total-DDT in agricultural, residential, and parkland soils (EPT, 1999). In aquatic environments, the TCEQ (2001) recommends sediment concentrations of 0.00354, 0.00142, and 0.00119 mg/kg dry weight, as ecological benchmarks for DDD, DDE, and DDT, respectively. For total-DDT, the TCEQ (2001) considers a concentration of 0.007 mg/kg dry weight as an ecological benchmark for sediments, while MacDonald *et al.* (2000), suggest a sediment TEC of 0.0053 mg/kg dry weight and a PEC of 0.57 mg/kg dry weight.

Neither of the two isomers of the metabolite DDD (*o,p'*-DDD and *p,p'*-DDD) were detected above the analytical detection limits in any of the samples collected from Grulla NWR or Muleshoe NWR (Tables 5 and 6). Of the two DDE isomers (*o,p'*-DDE and *p,p'*-DDE), only *p,p'*-DDE was detected above the analytical detection limits and only from one site at Grulla NWR (Site G06) and at two sites at Muleshoe NWR (Sites G07 and G10) (Tables 5 and 6). The DDT isomer, *o,p'*-DDT, was detected above the analytical detection limit at only one site, Site G11 at Muleshoe NWR, while the

isomer *p,p'*-DDT was also detected above the analytical detection limit at only one site, Site G06 at Grulla NWR (Tables 5 and 6), immediately down gradient of Arch Landfill.

The *p,p'*-DDE concentrations detected at Sites G06 (0.0011 mg/kg dry weight), G07 (0.00187 mg/kg dry weight), and G10 (0.0073 mg/kg dry weight) were less than all cited soil screening criteria (TAC, 1993; EPT, 1999; NMED, 2000). The *o,p'*-DDT level detected at Site G11 (0.00084 mg/kg dry weight) was less than the cited sediment screening criterion (TCEQ, 2001), while the *p,p'*-DDT concentration measured at Site G06 (0.00066 mg/kg dry weight) was less than the cited soil benchmarks (TAC, 1993; EPT, 1999; NMED, 2000). The sum of the detected metabolites and/or isomers of DDT (*p,p'*-DDD + *p,p'*-DDT) was calculated following Munn and Gruber (1997) to determine the total-DDT level for Site G06. The resulting value (0.00176 mg/kg dry weight) was less than all cited soil screening criteria for total-DDT (EPT, 1999; RAIS, 2002).

Polychlorinated Biphenyls

Results of the total polychlorinated biphenyl (total PCB) analyses for the six soil samples collected from Grulla NWR and the six soil/sediment samples collected from Muleshoe NWR are presented in Table 7. Every sample analyzed contained detectable amounts of PCBs. All results are presented in mg/kg dry weight.

Table 7. Results of total polychlorinated biphenyl (Total-PCBs) and total petroleum hydrocarbon (TPH) analyses in mg/kg dry weight for soil samples collected from six sites at Grulla National Wildlife Refuge (G01 - G06) and soil/sediment samples collected from six sites at Muleshoe National Wildlife Refuge (G07 - G12), 2003.

Sample Site	Total-PCBs	Detection Limit	TPH	Detection Limit
G01	0.00233	0.00157	32.90	1.30
G02	0.00561	0.00136	41.00	1.10
G03	0.00301	0.00156	28.90	1.25
G04	0.00543	0.00137	63.20	1.11
G05	0.00280	0.00134	24.90	1.08
G06	0.00461	0.00129	40.60	1.05
G07	0.00474	0.00183	63.80	1.48
G08	0.00486	0.00144	27.50	1.16
G09	0.00392	0.00167	33.40	1.35
G10	0.00891	0.00133	74.60	1.07
G11	0.05500	0.00178	106.00	1.43
G12	0.00410	0.00168	42.80	1.37

[Total Polychlorinated Biphenyls (PCBs)] First developed in 1929, polychlorinated biphenyls (PCBs) were used extensively in electrical transformers, capacitors, heat transfer fluids, and electrical utilities as lubricants, insulators, and coolants until production ceased in 1977 due to potential adverse environmental and human health affects (USEPA, 1994; Moring, 1997; ATSDR, 2000b). Total PCBs represent a quantification of approximately 209 individual congeners (Moring, 1997). These congeners are relatively stable compounds that exhibit low water solubilities, high heat capacities, low flammabilities, low electric conductivities, and low vapor pressures (USEPA, 1994;

Moring, 1997). Polychlorinated biphenyls are not naturally occurring and when released into the environment, degrade very slowly (ATSDR, 2000b). Reported half-lives for PCBs in lentic systems can range from 4 to 60 years (Spectrum, 2003).

In wildlife, PCBs can be teratogenic and tumorigenic and demonstrate a trend to bio-accumulate and bio-concentrate. In soils, the ecological screening benchmark recommended by the USEPA is 0.01 mg/kg, while the Oak Ridge National Laboratory considers a soil-total PCBs concentration of 40 mg/kg as a benchmark value protective of plants (RAIS, 2004b). The CCME recommends a soil-total PCBs concentration of 0.3 mg/kg as the screening criterion for agricultural, residential, and parkland soils (EPT, 1999). Buchman (1999), reports a soil-total PCBs concentration of 0.5 mg/kg dry weight as the target value for remedial efforts in agricultural areas and a concentration of 5 mg/kg dry weight as the target value for remedial activities in urban park and/or residential soils, while the State of Texas considers a soil-total PCBs concentration of 10 mg/kg, as protective of human health in residential areas (TAC, 1993). In sediments, the OME suggests a LEL of 0.07 mg/kg dry weight and a SEL of 5.3 mg/kg dry weight (Persaud *et al.*, 1993), while MacDonald *et al.* (2000), recommend a TEC of 0.06 mg/kg dry weight and a PEC of 0.68 mg/kg dry weight.

Measured soil-total PCB concentrations at Grulla NWR ranged from 0.0023 mg/kg dry weight at Site G01 to 0.0056 mg/kg dry weight at Site G02 (Table 7). In comparison, the detected soil-total PCB levels at Muleshoe NWR ranged from 0.0039 mg/kg dry weight at Site G09 to 0.0089 mg/kg dry weight at Site G10 (Table 7). The mean soil concentration calculated from the samples collected at Grulla NWR (\bar{x} = 0.004 mg/kg dry weight; n = 6) was less than the mean soil concentration determined for the soil samples taken from Muleshoe NWR (\bar{x} = 0.006 mg/kg dry weight; n = 4). All of the measured total PCB concentrations in soils collected from both refuges were less than the most conservative ecological screening criterion cited above (RAIS, 2004b).

Sediment collected from Site G11 contained a total PCB concentration of 0.055 mg/kg dry weight, while the sample taken from Site G12 contained a total PCB concentration of 0.0041 mg/kg dry weight. The total PCB concentration measured at Site G12 was less than all cited criteria, whereas the total PCB level detected in sediment from Site G11 approached the lower ecological threshold suggested by MacDonald *et al.* (2000), but was well less than the upper sediment screening values proposed by Persaud *et al.* (1993) and MacDonald *et al.* (2000).

Petroleum Hydrocarbons

Results of the total petroleum hydrocarbon (TPH) analyses for the six soil samples collected from Grulla NWR and the six soil/sediment samples collected from Muleshoe NWR are presented in Table 7. Every sample analyzed contained detectable TPH concentrations. All results are presented in mg/kg dry weight.

[Total Petroleum Hydrocarbons (TPH)] Total petroleum hydrocarbons (TPH) refers to the sum of total purgeable and extractable petroleum hydrocarbons present in a given sample medium (CCME, 1997). The TPH analysis includes several hundred hydrocarbons of petroleum origin that

can be broadly categorized as aliphatic and aromatic hydrocarbons (TCEQ, 2001). This analysis is a relatively inexpensive screening mechanism that is useful in determining the possible presence of petroleum contamination (TCEQ, 1995). For example, sediment collected by the USFWS from a stream at Hagerman National Wildlife Refuge (Grayson County, Texas) in 1999 believed to be impacted from a crude oil release had a TPH concentration of 2,455 mg/kg dry weight, whereas sediment collected from the same stream up gradient of the area affected contained a TPH level of 72 mg/kg dry weight (Giggelman and Bocanegra, 2000). However, it should be noted that without the knowledge of the source or composition of the petroleum contaminant, TPH in itself does not indicate an ecological risk of toxicity (NEPC, 2003).

In the U.S., national remedial standards have not been established for TPH levels in soils and/or sediments due in part to the lack of conclusive and supportive toxicity data (TCEQ, 1999). Some states, such as Kansas, Louisiana, and Utah, support TPH-remedial action levels based on the gasoline, diesel, and/or waste oil content present in a petroleum release, whereas other states, including Arizona, Arkansas, New Mexico, and Texas, base remedial target values on the toxicity of individual hydrocarbon compounds associated with petroleum releases, such as PAHs (polycyclic aromatic hydrocarbons) and BTEX (benzene, toluene, ethylbenzene, and xylene) (AEHS, 2004). The State of Colorado considers a TPH concentration of 500 mg/kg as the threshold value for subsurface and surficial soils, while in Missouri, dependent on certain parameters including soil characteristics and groundwater conditions, the Department of Natural Resources (MDNR) reports TPH concentrations ranging from 50 to 1,000 mg/kg as remedial goals for contaminated soils (AEHS, 2004). In Oklahoma, the Department of Environmental Quality (ODEQ) considers a TPH concentration of 50 mg/kg as an appropriate remedial target value for soils in areas where there are no impacts to surface water and/or groundwater from petroleum contamination (ODEQ, 2003). In Canada, the CCME (1997) reports a TPH value of 100 mg/kg or less as an acceptable level in soils at sensitive sites. The CCME (1997) defines sensitive sites as areas where there exists an imminent threat to public health or safety; all residential and agricultural areas; areas which have the potential of contaminating private, municipal, and/or industrial water supply sources; and areas within the boundaries of a protected water supply or ecological reserve. In Australia, TPH screening levels are based on the carbon chain fractions measured in the petroleum release (NEPC, 2003). Total petroleum hydrocarbon levels of 100 to 500 mg/kg are considered protective when carbon (C) chains range from C₆ to C₉, while TPH values ranging from 500 to 2,000 mg/kg are acceptable when the measured carbon fraction ranges from C₁₀ to C₁₅ (NEPC, 2003). The Australian National Environment Protection Council (NEPC) (2003) states that these TPH values apply to relatively recent releases. For aged and weathered petroleum hydrocarbons, the NEPC (2003) reports that the TPH thresholds for ecotoxicological effects may be significantly higher.

Measured soil-TPH concentrations at Grulla NWR ranged from 24.9 mg/kg dry weight at Site G05 to 63.2 mg/kg dry weight at Site G04 (Table 7). In comparison, the detected soil-TPH levels at Muleshoe NWR ranged from 27.5 mg/kg dry weight at Site G08 to 74.6 mg/kg dry weight at Site G10 (Table 7). With a mean concentration of 39 mg/kg dry weight (n = 6), the overall TPH level detected at Grulla NWR was less than the mean concentration measured at Muleshoe NWR (\bar{x} = 50 mg/kg dry weight; n = 4). Although the soil TPH concentration detected at Site G04 at Grulla NWR,

as well as the levels measured at Sites G07 (63.8 mg/kg dry weight) and G10 at Muleshoe NWR, exceeded the value cited by the ODEQ (2003) and the lowest remedial level reported by the MDNR (AEHS, 2004), they were all less than the CCME (1997) level recommended for sensitive areas and do not appear to represent a concern for ecological resources inhabiting either refuge.

Since definitive sediment-TPH screening criteria are not currently available, the sediment data from Muleshoe NWR were compared to the cited soil screening values (CCME, 1997; NEPC, 2003; ODEQ, 2003; AEHS, 2004). The sediments collected from Sites G11 and G12 contained TPH levels of 106 mg/kg dry weight and 42.8 mg/kg dry weight, respectively. The level detected at Site G12 was below all cited soil criteria (CCME, 1997; NEPC, 2003; ODEQ, 2003; AEHS, 2004). The concentration measured at Site G11 exceeded the ODEQ (2003) and CCME (1997) values and the lower thresholds reported by the NEPC (2003) and MDNR (AEHS, 2004), but was less than all other cited TPH criteria (NEPC, 2003; AEHS, 2004), and over 20 times below the value measured by Giggelman and Bocanegra (2000) in an area at Hagerman National Wildlife Refuge that was contaminated by a crude oil release.

Macroinvertebrates

A total of 2,387 aquatic macroinvertebrates representing 19 taxa were collected in the grab and dip net samples taken from Upper Paul's Lake at Muleshoe NWR (Table 8). This macroinvertebrate community was dominated by aquatic insects, with chironomids representing 67% of the entire sample. Aquatic macroinvertebrate trophic groups are presented in Table 9. Predators and collector-gatherers were the most common trophic groups collected at Upper Paul's Lake, while scrapers and filterers were rare.

The macroinvertebrate fauna collected from Muleshoe NWR is representative of the common taxa known from the saline lakes and playas of the High Plains region of Texas (Merickel, 1978; Kennedy *et al.*, 1998; Moorhead *et al.*, 1998; Hall *et al.*, 1999). Ephemeral aquatic habitats such as saline lakes, in the final stages of macroinvertebrate succession, are dominated by insect predators (Moorhead *et al.*, 1998). The dominance of predator species in Upper Paul's Lake most likely represents a climax stage of invertebrate community structure due to the perennial water source from groundwater. The healthy emergent vegetation and dense submerged vegetation within this lentic body also contributes to its macroinvertebrate assemblage.

CONCLUSIONS & RECOMMENDATIONS

The results of the metals analyses indicate that metals are present in surficial soils at higher levels at Muleshoe NWR than at Grulla NWR. Certain metals, primarily boron, exceed cited ecological screening criteria at Muleshoe NWR, but this may be attributed to the natural alkaline soils at the refuge. The results of the organic analyses indicate that residual organochlorine pesticides, total PCBs, and TPH are present in surficial soils at very low levels at both refuges. As with the metals,

Table 8. Macroinvertebrate taxa and their associated trophic relations collected from Upper Paul's Lake, Muleshoe National Wildlife Refuge, 2003 (Note - P = predator, CG = collector-gatherer, FC = filtering collector, SCR = scraper, and SHR= shredder).

Group/Family	Genus	Trophic Group	Number Collected
Anostraca		FC	6
Gastropoda		SCR	56
Hydracarina		P	1
Ephemeroptera	<i>Callibaetis</i> sp.	CG	340
Odonata	<i>Anax</i> sp.	P	6
	<i>Enallagma</i> sp.	P	219
	<i>Pantala</i> sp.	P	11
	<i>Tramea</i> sp.	P	17
Hemiptera	<i>Buenoa</i> sp.	P	50
	<i>Hesperocorixa</i> sp.	CG	4
	<i>Mesovelis</i> sp.	P	2
	<i>Notonecta</i> sp.	P	3
Coleoptera	<i>Berosus</i> sp.	P	61
	<i>Dineutus</i> sp.	P	5
	<i>Laccophilus</i> sp.	P	6
	<i>Tropisternus</i> sp.	P	1
	<i>Uvarus</i> sp.	P	1
Diptera	Chironomidae	P/CG/SHR	1595
	<i>Culex</i> sp.	FC/CG	2

Table 9. Number and percentage trophic groups of macroinvertebrate taxa represented in samples collected from Upper Paul's Lake, Muleshoe National Wildlife Refuge, 2003.

Trophic Group	Number Represented	Percentage of Sample
Scrapers	56	2
Collector-Filterers	7	0.003
Collector-Gatherers	877	37
Shredders	532	22
Predators	915	39.00

the organics detected at Grulla NWR were lower than the levels measured at Muleshoe NWR. Overall, the contaminants detected at both refuges were below concentrations where adverse effects to ecological resources would be expected to occur.

The macrobenthic community sampled at Muleshoe NWR appeared to be typical of lentic bodies within the region. No macrobenthic samples were collected from Grulla NWR for comparative purposes due to drought conditions.

Based on observations at the time sampling was conducted in conjunction with documentation of past disposal operations at the site, contamination is present at the Arch Landfill. However, this contamination does not appear to be migrating offsite in any significant amount down gradient into Grulla NWR. This may be attributed to the sparsity of rainfall in the area which reduces the likelihood of surficial contamination migrating continuously from the landfill through the arroyo into the Refuge. The possibility of contamination from the landfill entering the Refuge through stormwater runoff from a significant storm event still exists. Therefore, it is recommended that any future sampling be conducted within 24-hours after a major rainfall event to account for the possibility of contaminants being washed into Grulla NWR through stormwater runoff.

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APPENDIX A
(ANALYTICAL METHODS)

Method Code: 003 for % Moisture, % Dry Weight

Laboratory: Geochemical % Environmental Research Group, Texas A&M

Approximately 1 gram of wet sample is weighed into a clean, labeled, pre-weighed 10 ml beaker. The beaker is placed in a forced air oven at approximately 75°Celsius for 24 hours. The beaker with the dry sample is then weighed and the % dry weight is calculated by the formula:

$$\frac{(\text{wt. dry sample and beaker}) - (\text{wt. beaker})(100)}{(\text{wt. wet sample and beaker}) - (\text{wt. beaker})}$$

Method Code: 004 for 1,2,3,4-tetrachlorobenzene, 1,2,4,5-tetrachlorobenzene, aldrin, hexachlorobenzene (HCB), heptachlor, alpha hexachlorocyclohexane (α BHC), alpha (α) chlordane, beta hexachlorocyclohexane (β BHC), cis-nonachlor, delta hexachlorocyclohexane (δ BHC), dieldrin, endosulfan II, endrin, gamma hexachlorocyclohexane (γ BHC), gamma (γ) chlordane, heptachlor epoxide, mirex, o,p'-dichloro-diphenyl-dichloroethane (o,p'-DDD), o,p'-dichloro-diphenyl-trichloroethane (o,p'-DDT), oxychlordane, p,p'-dichloro-diphenyl-dichloroethane (p,p'-DDD), p,p'-dichloro-diphenyl-dichloroethylene (p,p'-DDE), p,p'-dichloro-diphenyl-trichloroethane (p,p'-DDT), pentachloro-anisole, toxaphene, trans-nonachlor, chlorpyrifos, and total polychlorobiphenyls (PCBs)

Laboratory: Geochemical % Environmental Research Group, Texas A&M

The soil/sediment samples were freeze dried and extracted in a Soxhlet extraction apparatus. Briefly, the freeze dried soil/sediment samples were homogenized and a 10 gram sample was weighed into the extraction thimble. Surrogate standards and methylene chloride were added and the samples extracted for 12 hours. The extracts were treated with copper to remove sulfur and were purified by silica/alumina column chromatography (MacLeod *et al.*, 1985; Brooks *et al.*, 1989) to isolate the pesticide and PCB fractions. The quantitative analyses were performed by capillary gas chromatography (CGC) with electron capture detector for pesticides and PCBs (Wade *et al.*, 1988). There are specific cases where analytes requested for the pesticide and PCB analyses and are known to co-elute with other analytes in the normal CGC with electron capture. These include the pesticide endosulfan I and the PCB congeners 114 and 157. In these cases, the samples were analyzed by CGC with mass spectrometer detector in the SIM mode.

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Method Code: 028 Analytical Methodology for Total Petroleum Hydrocarbons (TPH) in Sediments.

.0

Laboratory: Geochemical & Environmental Research Group, Texas A&M

Sediment samples were freeze dried and extracted in a Soxhlet extraction apparatus. The freeze dried sediment samples were homogenized and a 15.0-gram sample was weighed into the extraction thimble. Surrogate standards and methylene chloride were added and the samples extracted for 12 hours. The extracts were treated with copper to remove sulfur. Extract was then rotovaped to 5.0 ml and then brought to dryness under a clean nitrogen stream. GC internal standards were added and the extract was run on gas chromatograph with a flame ionization detector. TPH was determined by summing the total unresolved complex mixture (UCM) and the total resolved (all peaks in the chromatogram). The concentration was based on the average of the response factors for alkanes from n-C10 through n-C34.

Method Code: 006 for Soil/Sediment Clay, Silt, and Sand Grain Sizes

Laboratory: Geochemical % Environmental Research Group, Texas A&M

A small aliquot of sediment is treated with 30% hydrogen peroxide to remove organic coating from grains. A dispersing agent is then added to the sample. The sand/mud fractions are then separated using a 63 micron sieve. The sand fraction (greater than 63 microns) is retained on the screen and the mud fraction (silt and clay less than 63 microns) is washed into a 1 liter volumetric cylinder. The sand fraction is dried, sieved on a 63 micron screen and weighed. The sediment which passes through the screen a second time is added to the 1 liter cylinder. The mud fraction is analyzed by stirring the cylinder and sampling 20 ml aliquots at 4 and 8 phi intervals. The 4 and 8 phi samples are dried and weighed. The % sand, silt, and clay fractions are determined on a dry weight basis.

Method Codes: 001, 004, and 006 for aluminum, barium, beryllium, boron, cadmium,

chromium, copper, iron, lead, magnesium, manganese, molybdenum, nickel, strontium, vanadium, and zinc.

Laboratory: Research Triangle Institute

Homogenization (001) - Soil/sediment samples are pre-homogenized using a food processor. A portion of the sample is then freeze dried for determination of moisture content and ground to 100 mesh with a mill.

Digestion for Graphite Furnace and Cold Vapor Atomic Absorption (GFAA) Measurement (004) - Using a CEM microwave oven, 0.25 to 0.5 grams of freeze dried sample are heated in a capped 120 ml Teflon vessel in the presence of 5 ml of Baker Instra-Analyzed nitric acid for three minutes at 120 watts, three minutes at 300 watts, and 15 minutes at 450 watts. The residue is then diluted to 50 ml with laboratory pure water.

ICP (006) - ICP measurements are made using a Leeman Labs Plasma Spec 1 sequential or ES2000 simultaneous spectrometer.

Method Codes: 001, 004, and 007 for arsenic and selenium.

Laboratory: Research Triangle Institute

Homogenization (001) - Soil/sediment samples are pre-homogenized using a food processor. A portion of the sample is then freeze dried for determination of moisture content and ground to 100 mesh with a mill.

Digestion for Graphite Furnace and Cold Vapor Atomic Absorption (GFAA) Measurement (004) - Using a CEM microwave oven, 0.25 to 0.5 grams of freeze dried sample are heated in a capped 120 ml Teflon vessel in the presence of 5 ml of Baker Instra-Analyzed nitric acid for three minutes at 120 watts, three minutes at 300 watts, and 15 minutes at 450 watts. The residue is then diluted to 50 ml with laboratory pure water.

GFAA (007) - GFAA measurements are made using a Perkin-Elmer Zeeman 3030 or 4100ZL atomic absorption spectrometer.

Method Codes: 001, 004, and 008 for mercury.

Laboratory: Research Triangle Institute

Homogenization (001) - Soil/sediment samples are pre-homogenized using a food processor. A portion of the sample is then freeze dried for determination of moisture content and ground to 100 mesh with a mill.

Digestion for Graphite Furnace and Cold Vapor Atomic Absorption (GFAA) Measurement (004) - Using a CEM microwave oven, 0.25 to 0.5 grams of freeze dried sample are heated in a capped 120 ml Teflon vessel in the presence of 5 ml of Baker Instra-Analyzed nitric acid for three minutes at 120 watts, three minutes at 300 watts, and 15 minutes at 450 watts. The residue is then diluted to 50 ml with laboratory pure water.

Cold Vapor Atomic Absorption (CVAA) - mercury measurements are conducted using SnCl₄ as the reducing agent. A Leeman PS200 Mercury Analyzer is employed for the analysis.
